RANGELAND FIRE EFFECTS

A SYMPOSIUM

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SYMPOSIUM KEYNOTE

Wildland Fire: A Perspective

Clair M Whitlock

Fire has always been a powerful factor in shaping the vegetation in the sagebrush-grass communities. While lightning has historically been the primary source of ignition, man is now the significant factor. In fact, 70% of Idaho's fires are now man-caused. In the early days of public land management, wildfire was viewed in much the same way as a structural or personal property fire - a destructive force which must be fought with whatever resources were available. There was basis for this attitude, what with the catastrophic forest fires of the Lake States and the Northern Rockies, as well as range fires and chaparral fires throughout the Western United States. Early on, the land management agencies adopted a policy of manning to contain fires by 10:00 a.m. of the day following ignition, usually without regard to resource values. We all pitched in and supported the Smokey Bear Campaign to stop man-caused fires and, in our fervor, we continued stomping out all fires at the smallest size possible considering available capability.

As we gained a rudimentary understanding of the relationship of fire to the vegetation, we began to experiment with what were termed "controlled burns." All too often they really were either uncontrolled or would not burn at all. Of equal concern, we often failed to get the vegetative response we expected. These are symptoms of lack of knowledge in either fire behavior or in response of vegetation to a fire event.

As we have gained more experience and knowledge, our attitude toward fire has changed. With this change, the agencies have slowly shifted from a strategy oriented strictly to fire suppression to one of fire management. The concept of fire management is easier to describe than to implement, as we all have our biases and tend to resist change. Fire management recognizes several things: (1) fire is an integral part of the natural scheme of things in our wildlands; (2) people do start intentional or accidental unwanted fires; (3) Mother Nature also ignites numerous fires; (4) we can intentionally start fires to accomplish some wanted positive effect or goal; (5) we have limited resources for managing fire and vegetation, thus we must determine priorities and devise cost-effective techniques and strategies commensurate with our capability; and (6) we have a knowledge base which will guide us in setting priorities and developing cost-effective strategies.

At this symposium we want to expand that knowledge base, primarily as it applies to the sagebrush-grass ecosystem of the Northern Great Basin/Snake River country. We will look at the impact of fire not only on vegetation but also on wildlife habitat, soil, and air.

We are purposely not delving into fire-related problems in the annual grass and weed vegetative types. That is a broad enough subject for a separate symposium. I think, however, you will pick up some useful information for rehabilitating these annual grassland situations.

We also have purposely designed the symposium to not provide cookbook fire prescriptions. This would require broadening the subject matter into fire behavior and weather considerations, diluting our objective of dealing with fire effects or impacts. You will be given some tools and knowledge which could be the foundation for your future prescriptions.

As we have searched for more knowledge and answers in connection with BLM's fire management program, we find a general lack of published data on fire relationships in the sagebrush-grass community. At the same time, we know there is research and study being done. Furthermore, we know many of you are doing prescriptive burning as you attempt to improve forage production and quality, improve wildlife habitat, and reduce fuel loads to prevent catastrophic fires. The planned prescriptions for BLM alone in 1984 involved 56 projects spanning 67,600 acres, including 47,000 acres in Idaho. I am certain you participants have a powerful mass of knowledge in your grasp. It is critical that you share that knowledge. The presentors, in addition to providing information, will serve as catalysts to create a dialogue. You will decide the success of the symposium through your participation and interaction. Perhaps the most important result of your being here will be to develop a network of researchers, practitioners, and land managers who share a common interest in the role of fire in wildland management. Such a network will be invaluable in sharing and spreading experience and data on fire and vegetation responses. I urge you to go for it.

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Fire in the Sagebrush-Grass Ecosystem--The Ecological Setting

A. H. Winward

Abstract

Sites within the sagebrush-grass ecosystem may react to burning quite differently since they do have unique environments, different species composition, and different fuel loading characteristics. We need to put more effort into matching prescribed burns to specific kinds of sagebrush sites.

Introduction

A trip into the Boise area, either by car or by air soon reminds one of the vast acreage covered by sagebrush (Artemisia). If we were to get in a car and travel westward from Boise we would be driving through sagebrush grassland for several hours, until we finally reached the eastern slopes of the forested Cascade Mountains. Traveling north we would encounter islands of sage intermittently among forest or other rangeland vegetation well into the southern portions of British Columbia. Eastward the gray-haze of sagebrush would be discernible through all of Idaho, sections of Montana or Wyoming, and extending into portions of several Plains states. And, southward would be just plain sagebrush monotony the full length of Nevada, finally giving way to the cactus and mesquite of the warmer true desert climates of the southwest.

Do you realize that every one of the far western states has some acreage of woody (shrubby) sagebrush present? And, if we were to add the herbaceous Artemisias we would probably include all of the states in the Union including Alaska and Hawaii. There are nearly 100 million acres of shrubby sagebrush. This is enough to entirely cover the ten most northeasterly states with enough leftover to add a little spice and aroma to 4-5 more. We have a lot of sagebrush in our western United States.

Sagebrush Historically

Apparently we have always had a lot of sagebrush. Yensen (1980) made reference to an article by Elliott (1913) who claimed the Indian name for the Snake River was "Pohagwa" meaning "Sagebrush River." In 1839 a young naturalist named John Townsend recorded that "the Snake River Plain was covered with rugged lava and twisted wormwood (sagebrush)." Fremont (1845) called the region the "Sage Desert" because it was "covered with Artemisia as far as the eye could see."

Sagebrush did not come into the west with the railroad as I was told as a young boy in southeastern Idaho. Sagebrush seed did not come into the area in the wool of sheep and hair of horses and cattle as some would have you believe. Instead all the kinds of sagebrush seed were already here at the arrival of those first "outsiders" in the early 1800's. I would submit further that it had already been here long enough by this early settlement period to have adapted itself to a variety of ecological situations; long enough, in fact, to have developed new species and forms capable of surviving many different environmental situations. By the time European people came into the scene sagebrush had already survived several small ice ages, many volcanic eruptions, uplifting of mountain ranges, numerous wet and dry cycles and, yes, numerous range fires. No wonder it has adapted, and is continuing to adapt to so many ecological situations.

Sagebrush Adaptations

The genus Artemisia is very interesting as one looks at its variations. We are fast approaching recognition of 25 kinds of shrubby Artemisias in the western U.S. Within these 25 we find adaptations to sites ranging from 7 to 25 inches of annual precipitation. Artemisias occur from the valley floor to subalpine (300 feet to above 11,000 feet elevation). We find them on well-drained soils as well as flooded playas; on deep, arable sites to extremely shallow, rocky sites; in areas with long growing seasons to places where there essentially is no frost-free period.

It is interesting that in all these ecological situations sagebrush has relatively few natural enemies. Insects seldom kill vast acreages of sagebrush. There are no bark beetles or leaf bores which wipe out extensive populations as occur in other important western genera. Instead, insects such as the Aroga moth set back or occasionally kill a few acres of sage which are most often rapidly reclaimed by new sagebrush seedlings.

Seldom is sagebrush grazed out. Only occasionally do we find areas where overwintering livestock or wild ungulates browse species of this genus to the extent of elimination. In fact, many of our current grazing activities tend to stimulate sagebrush reproduction and growth by reducing some of the competing associated species. Overall, in relation to the amount of sagebrush in the west, a relatively small portion is consumed as forage by all users including native ungulates and insects.

Sagebrush and Fire

One natural phenomenon to which sagebrush is often subjected is fire and that is why we are here today. Generally, most sagebrush

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species are highly susceptible to fire. A few taxa have special adaptations and are exceptions. Why would a genus which has adapted itself so well for surviving harsh climates have failed to develop a better tolerance for fire? Or has it? Perhaps we should define tolerance. If we mean ability to withstand temperatures generated in our rangeland fires, it is not very tolerant. Exceptions would be those few with ability to resprout after burning but even with these mortality is often high in many fires. However, if by tolerance we are referring to ability of a species to maintain itself on a long-term basis in spite of periodic burning then perhaps sagebrush should be considered rather fire tolerant. In your experiences, how long does fire normally remove sagebrush from a site. In most cases it is well on its return to the site 5-10 years after a burn. Normally enough sagebrush seed remains in the soil surface for rapid recolonization.

Occasionally we have exceptions to this where 2-3 repeated fires on the same area kill new young plants before they have ability to produce more seed to be stored in the soil. We have a good example of this around the Boise area. However, we have a rather unnatural situation here where cheatgrass prominence and frequent natural as well as unnatural ignitions have increased fire frequency to an almost yearly event. No woody species, not even the rabbitbrushes, can survive this situation. Normally sagebrush survives fires through rapid regeneration of seedlings and in this sense it may be called fire tolerant.

Those few sagebrush species which have an ability to resprout from root crowns or lower stem bases after being top-killed include threetip (A. tripartita) and the three subspecies of silver sagebrush (A. cana). However, even with these only a portion survive many of our range fires. Just recently we found that a variation of A. tridentata in the high elevation of southeastern Idaho and adjacent portions of Wyoming, Colorado, and Utah has an ability to resprout after burning. Thirty to fifty percent of these plants are surviving prescribed burns. This is rather significant since this is the first time we have encountered a resprouting member of big sagebrush. This plant has been erroneously referred to by several names but has recently been found to belong to the taxon "spiciformis" (Goodrich et al., in press). We have found that most material assigned to spiciformis the past 15 years is in error.

We must consider this feature of sprouting in plans to thin or remove sagebrush. Fire may not always be the best approach to accomplish our purpose when these or other sprouting shrub species occur. However, most of the kinds of sagebrush we presently prescribe burn do not resprout; i.e., mountain big sagebrush (A. ssp. vaseyana), basin big sagebrush (A. ssp. tridentata), and Wyoming big sagebrush (A. wyomingensis).

What has been the historical role of fire in the sagebrush-grass ecosystem? We often, almost casually, make such statements as "Fire was always a part of the sagebrush region" or "Fire is a natural component of the Sagebrush Region." We recently defend use of fire as a manipulative tool because it is "natural." I doubt that anyone would disagree that fire has played a significant role in vegetation succession in this ecosystem. Records of early travelers, as well as burn scars on adjacent trees and just plain common sense, indicate that it has been an important environmental feature along with climate. But, what do we know of historical fires; their intensities, sizes, and effects. How should this influence our fire prescriptions today?

The literature suggests that fire frequencies have averaged from 32 to 70 years in sagebrush-grass communities (Wright et al. 1979) or perhaps ss frequent as 20-25 years (Houston 1973). In contrast, fire frequencies in coniferous forest communities may range from 50-250 years (Arno and Davis 1980; Sneck 1977). If we use our most current knowledge of habitat types within the sagebrush ecosystem we find a variety of fuel situations. If we couple this with yearly climatic variations, terrain and storm track patterns no wonder there is a natural wide variation in fire frequencies. I would expect that as we increase our understanding of the ecology of different plant communities and habitat types we will be able to develop better site-specific prescriptions for our burns.

Historically fire likely has had little or no influence on vegetation on some habitat types. Black sagebrush (A. nova) and low sagebrush (A. arbuscula) in best of conditions have marginal fuel situations for carrying fire. On these sites fire has probably always been a rare occurrence. The same may be true for some Wyoming big sagebrush habitat types. Fire literature indicates that it is difficult to burn sagebrush areas unless there is at least 600-700 lbs/acre (674-786 kg/ha) of herbaceous fuel present (Beardall and Sylvester 1976). Some of our drier Wyoming big sagebrush types do not produce this much herbaceous material even without grazing present. Literature also suggests burning should not be considered unless sagebrush cover exceeds 20 percent (Pechanec et al. 1954). Again some sagebrush taxa may not reach these cover values and so probably have had little fire influence historically. On the other hand several taxa of sagebrush, especially mountain big sagebrush, have fuel situations and shrub cover values which normally result in easy burns. These sites have likely had a long history of fire and would fit the 20 to 30 year frequencies discussed earlier. Champlin and Winward (1980) found that germination of soil-stored seed of mountain big sagebrush is actually stimulated by fire which would support a history of development under frequent burning.

In addition to natural fire frequencies, one could assume fire intensities likewise could be somewhat variable on different sites. We sometimes feel that historical fires were good because they were natural when, in fact, some probably had severe impacts on sensitive species such as Idaho fescue (Festuca idahoensis) and bitterbrush (Purshia tridentata) or on many shallow rooted, non-rhizomatous forb species (Pechanec et al. 1954). They likewise sometimes had serious effects on native wildlife species because of their far-reaching boundaries.

I believe it would be safe to conclude that different sites in the sagebrush-grass ecosystem may potentially react to burning quite differently since they do have different environments, different species composition, and different fuel loading characteristics. It would probably be good if we would put more effort into matching our prescribed burns to specific kinds of sagebrush sites. I see more literature each year where this is being done, especially in timbered situations (Fischer 1981).

Concerns

I will spend the remainder of my time presenting some basic concerns I have relative to prescribed burning in sagebrush-grass communities. These thoughts are not necessarily backed by scientific data. Some come from general observations I have made reviewing results of prescribed burns or from conclusions I had reached reading pertinent literature.

1. FIRE EMPHASIS:

There has become a rather expanded interest in use of fire as a manipulative tool in sagebrush communities--almost to the exclusion of use of mechanical or chemical approaches. The general basis for fire emphasis is economics and environmental concerns. While I do not object to this emphasis, I wonder if we do not short change ourselves by overemphasizing any one method for manipulating sagebrush. To me, prescribed fire should be a tool for sagebrush removal not necessarily the tool.

There are many acres of land which presently are in need of some type of sagebrush thinning process which are not suited to practical prescribed burning. These areas, which do not fit the herbaceous or shrub cover values due to environmental limitations, will not be treated even though they are in extremely poor ecological condition. If not treated in some way, there will be no opportunity for them to grow a balanced understory in our lifetime. If these areas are to be neglected, let it be on the basis of some factor other than strict reliance on fire as our only manipulative tool.

2. OBJECTIVES:

I believe we need to define our objectives in more quantifiable terms when we develop prescribed fire plans. Some fire projects are proposed to "improve wildlife habitat." Sometimes the wildlife species are not even defined let alone the habitat features that are to be improved.

Others burn to "increase livestock forage." What forage? Idaho fescue is forage but then so is cheatgrass (Bromus tectorum). Which are they after? In precise terms--what are the specific objectives to be accomplished? When is a burn successful?

3. MONITORING:

I believe we need to be more careful and precise in monitoring fire effects. In fact, I believe we should probably be doing more monitoring of fire treatments. I am not just referring to researchers. I am including those who are doing the on-the-ground work. I have seen individuals that are excited that they have doubled the height of bluebunch wheatgrass (Agropyron spicatum) yet, they failed to look at what happened to ground cover. Too many people get excited over a good kill of brush yet forget about what has actually happened to the entire community. During the years 1982 and 1983, I looked over more prescribed burns which I would classify as failures, in the sense of what happened to the total vegetation or site, than those that appeared successful. If we do not have good monitoring follow-up, we will tend to see only what we want--which may be rather biased. I prefer to see us spend a little more time measuring several parameters used for monitoring communities or site changes rather than putting our effort into just pounds of forage or acres of killed brush.

4. RABBITBRUSHES:

There often is a perception that presence of rabbitbrush on a site is an automatic "red flag" against treating that site. This fear is based on the idea that rabbitbrush will be released and take over the site at the expense of more desirable species. On some sites there is good cause for concern because rabbitbrush can resprout and/or produce abundant seed and become seriously competitive. However, under some situations I feel that they are not the problem plants some have claimed them to be and we should not necessarily view their presence as a restriction against treatment. My reasons include:

a. Rabbitbrushes occur as many species and subspecies; some with unique requirements for management. For example, both Chrysothamnus nauseosus ssp. albicaulis and C. nauseosus ssp. hololeucus are gray in stem and leaf color, relatively valuable as forage species (McArthur et al. 1979) and apparently relatively easy to manage (i.e., susceptible to chemicals and fire). Compare these with C. nauseosus ssp. consimilis and C. nauseosus ssp. graveolens which both are green/gray in color, generally unbrowsed, and apparently relatively difficult to kill with chemicals or fire.

We need to identify the rabbitbrushes to the subspecies level where appropriate and begin assembling a record of best management approaches by subspecies. Publication of the proceedings from a symposium on sagebrush and rabbitbrush held in Provo, Utah (1984) is in progress in which a taxonomic key and geographic maps for each kind of known rabbitbrush in the western U.S. is presented (Anderson, In press). These proceedings will be available within the next year and should help us properly identify and manage rabbitbrush by subspecies.

b. Under most situations in the Intermountain and Pacific Northwest Regions, the rabbitbrushes are successional to other perennial shrub species. As such, they generally are relatively short-lived shrubs and may have less detrimental impact, on a long-term basis, than has been assigned to them. Additionally, they are less competitive with associated grasses and forbs than sagebrushes since they extract much of their moisture and nutrients from deeper in the soil profile (Frischknecht 1963). As long as we continue to emphasize maintenance or establishment of other perennial species on sites following treatment, some presence of rabbitbrush may be less serious than we have tended to believe. In fact, presence of some rabbitbrush may be beneficial to associated plant species by cycling nutrients from deep soil horizons to the surface where they can be utilized by other species. Additionally, rabbitbrush crowns can provide moderating situations (microsites) suitable for easier establishment of new seedlings of some grass and forb species providing rabbitbrush densities are not too high.

We still need to use caution when burning areas where rabbitbrush has potential to dominaie the site after treatment. But, we should remain somewhat open-minded to the possibility that this will not happen everywhere rabbitbrush is present prior to treatment. We need to identify those situations and those taxa of rabbitbrush which potentially can cause us difficulties with management.

I certainly do not want to discourage use of fire as a management tool on our wildlands. When someone asks me what approach to use to reduce shrub cover my thoughts are first fire, followed by comparison with other potential approaches.

Fire was a natural part of many sagebrush grassland communities and it should remain so. I hope we keep its use in the realm of a science and art not just a perceived, easy, inexpensive approach to management.

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Fire in Sagebrush-Grass Ecosystems: Successional Changes[1]

Stephen C. Bunting[2]

Abstract

Effects of fire on sagebrush-grass vegetation are reviewed with respect to successional changes in community biomass production, plant cover and plant density. The response of the three parameters to fire and successional processes varies. Some aspects of community structure are greatly affected by fire and others change little. This variation in response can alter the perceived successional stages measured after fire and the ability to meet some land management objectives with the use of prescribed fire.

Introduction

Fire acts as a disturbance factor to the vegetal development of most plant communities. When prescribed fire is used as a vegetation manipulation technique in land management, the manager is using secondary succession to meet specific objectives. This requires an understanding of the successional processes that occur after fire. Succession has been described by Daubenmire (1968) as any unidirectional change that can be detected in the proportions of species in a stand or the complete replacement of one community by another. Various other definitions exist but most are similar in many respects to that of Daubenmire's.

This paper will concentrate on a portion of Daubenmire's definition, "change that can be detected in proportions of species," as it applies to the effect of fire on sagebrush-grasslands. This is important to the understanding of secondary succession because the type of measurement used to detect changes in proportions will affect the eventual successional sequence developed.

Researchers have employed a number of parameters to describe successional change in sagebrush-grass vegetation following fire. Current annual production was used by Blaisdell (1953), Pechanec et al. (1954), Mueggler and Blaisdell (1958) and Harniss and Murray (1973). Canopy cover of shrubs (Young and Evans 1974), basal cover of herbaceous species (Peek et al. 1979), density (Pechanec and Hull 1945, Mueggler and Blaisdell 1958, Bunting et al. In Press) and plant frequency (Peek et al. 1979, Kuntz 1982) can all be found in the literature as measures of succession after fire in sagebrush vegetation.

In addition, parameters which may directly or indirectly affect species composition have been studied. These include plant mortality (Wright and Klemmedson 1965, Bunting et al. In Press), measures of reproductive potential (Young and Evans 1974, Uresk et al. 1976, Young and Miller In Press) and morphological characteristics of plant species (Wright 1971, Young and Evans 1974).

The use of these parameters has resulted in what appear to be different responses of sagebrush-grass vegetation to fire. However, predictable trends become evident, if one views the population parameters in relation to one another. It is also necessary to stratify the vegetation into classification units in order to identify general trends.

Discussion

In the following discussion three basic parameters of plant communities will be reviewed. They include annual biomass production, coverage (basal and foliar) and plant density. These are the most common parameters reported in the literature and are also the most easily visualized. Annual biomass production is the parameter most directly related to animal carrying capacity. As a result, increases in some component of production is usually included as an objective in prescribed fire projects. Coverage is an important consideration because the areas formerly occupied by sagebrush become available to other plants after the fire. In order for these areas to be occupied, the establishment of new individuals is necessary. This is dependent upon changes in density.

Annual biomass production

Artemisia is a major component of sagebrush-grass vegetation. The initial response of this vegetation to fire is a shift of the resources utilized by *Artemisia* to the herbaceous species present on the site at the time of the fire. In most instances, total biomass production on the site is reduced, as found by Harniss and Murray (1973). Total production does not return to preburn levels until the re-establishment and dominance of *Artemisia* occurs. There is an increase in the biomass production of other components of the community. Biomass production of perennial grasses has been reported to increase up to 24% the first year after the fire (Uresk et al. 1976). Most studies, however, indicate that two to several years may be required before grass production

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[1] In this paper *Artemisia tridentata* subsp. *tridentata*, *A. tridentata* subsp. *wyomingensis* and *A. tridentata* subsp. *vaseyana* will be referred to as *A. tridentata*, *A. wyomingensis* and *A. vaseyana*, respectively.

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increases to preburn levels (Blaisdell 1953, Mueggler and Blaisdell 1958, Peek et al. 1979). After this recovery period, grass production on the burned area may exceed that of the unburned area by over 100% (Harniss and Murray 1973, Peek et al. 1979). The period of increased productivity varies greatly by habitat type and may last for over 20 years (Harniss and Murray 1973).

The time required for increased grass production to occur depends upon the grass composition present at the time of the burn and the climatic conditions at the time of and following the fire. Grass species such as Agropyron spicatum, Poa sandbergii, Elymus cinereus and Koeleria pyramidatus are usually resistant to fire. Grass components dominated by these species will likely respond more rapidly than those dominated by fire sensitive species such as Festuca idahoensis, Stipa thurberiana and S. comata.

Climatic conditions at the time of the fire may increase the severity of effects on plants. This is especially important with the more fire sensitive species. For example, Wright and Klemmedson (1965) reported that the mortality of Stipa thurberiana and S. comata varies from 0 to 100% depending upon the season and temperature of the fire. Other plants may survive the fire but have reduced vigor for many years.

Change in perennial forb productivity is less variable than that of the perennial grasses. Most of the forbs present in sagebrush-grass vegetation are active only during the early part of the growing season and are dormant when most fires occur. The buds are well below ground level and the plants accumulate litter slowly. Consequently, most are tolerant of all but the most severe fires. Once released from the competition of the Artemisia, these species respond quickly. Their production is eventually reduced as the perennial grasses regain vigor and the Artemisia seedlings re-establish on the site (Mueggler and Blaisdell 1958, Harniss and Murray 1973).

The production of annual plants may vary widely depending upon the habitat type and the ecological condition of the site. Areas dominated by Artemisia wyomingensis or A. tridentata frequently contain Bromus tectorum in the understory. Bromus tectorum will rapidly invade those areas formerly occupied by Artemisia unless substantial perennial herbaceous cover is present on the site. Other annuals which may respond similarly include Lactuca serriola and Sisymbrium altissimum. Annual production on these sites may increase several fold and remain high for many years.

On cooler and more mesic sites, such as those dominated by Artemisia vaseyana, annual production is usually less on more arid sites. Introduced annuals, such as those mentioned previously, are not as common nor as competitive on all sites. Native annuals such as Collomia linearis, Collinsia parviflora and Stellaria spp. increase on the burned sites but are rapidly suppressed by perennial plants.

The immediate response of shrub production on burned sites depends upon the amount of sprouting shrubs such as Chrysothamnus and Tetradymia canescens present at the time of the fire. These plants not only resprout readily but also establish easily from seed. Production on burned areas may exceed that of unburned areas within 3 years after the fire and remain high for 20 years (Chadwick and Dalke 1965, Harniss and Murray 1973). These plants begin to decline in vigor after this time if they are not reburned. Post-fire productivity of Artemisia depends upon the species involved. Artemisia tripartita resprouts in parts of its range and may rapidly re-establish on a site (Pechanec et al. 1965). Artemisia vaseyana establishes readily from seed (Winward 1970) and also reaches reproductive maturity early. Artemisia wyomingensis re-establishes more slowly and production may be reduced for 50 years or more. When frequent fires remove the seed source of nonsprouting plants, productivity in the community may remain low for many years.

Plant Density and Cover

Changes in density and cover of herbaceous species occur as rapidly as changes in productivity. The increased production of perennial grasses during the first 3 years after fire can be primarily attributed to the release from competition by Artemisia and not to increases in plant density. Total grass cover is reduced by fire (Wright and Klemmedson 1965, Uresk et al. 1976). This is due to a reduction in plant density and vigor. Fire resistant species such as Agropyron spicatum may have slight or no decreases in plant coverage and recover quickly (Uresk et al. 1978, Peek et al. 1979, Kuntz 1982). Festuca idahoensis, however, is more fire sensitive. Severe fires caused a 27% reduction in density and a 50% reduction in basal cover of this species (Conrad and Poulton 1966). Countryman and Cornelius (1957) reported a 80% reduction in basal cover after a wildfire in California. The basal cover of Festuca may be reduced over 30% on low severity fires even when mortality is low (Wright et al. 1979).

Increases in plant density due to perennial grass establishment is slow and not well understood. Only slight increases in Agropyron and Festuca have been observed within the first 5 years on many fires (Young and Evans 1978, Peek et al. 1979, Kuntz 1982). It appears that the increased numbers of plant inflorescenses observed after fire (Wright and Klemmedson 1965, Uresk et al. 1976, Young and Miller In Press) do not necessarily result in new plants becoming established.

Annual plants are well adapted to occupy space released by disturbance and will rapidly dominate the area if perennials are not present (Piemeisel 1951, Wright and Klemmedson 1965). Density will increase during the first 2 to 3 years (Young and Evans 1978) and then remain relatively constant for a period of years until Artemisia re-establishes. Burning prior to seed shatter of Bromus tectorum can reduce the density by 90% as compared to burning later in the fall (Pechanec and Hull 1945). The effect is short-lived, however, and Bromus density returns to unburned levels within 2 years. Bromus is a winter annual and germinates as soon as fall moisture is available (Hull and Hansen 1974). Consequently, it is 6 weeks ahead of Agropyron in the spring (Tisdale and Hironaka 1981). Fall germination and high densities make it extremely competitive with perennial grass seedlings. High populations of Bromus and possibly other annuals seriously reduce or prevent seedling establishment of perennials. Artemisia seedlings may become established but the flammability of the annual grass creates an excellent fuel bed for successive fires which may prevent their survival (Klemmedson and Smith 1964).

Perennial forbs are more resistant to fire than grasses in sagebrush-grasslands, and most are not harmed by late summer or fall fires (Wright et al. 1979). The response is primarily increased productivity, however, and not increased density (Kuntz 1982, Bunting Unpub. data). Basal area may increase due to greater vigor of the plant. Species such as Lupinus sericeus and Balsamorhiza sagittata often become visual dominants after the fire and certainly produce more biomass than unburned plants. Increased densities within the first few years after the fire have not been observed (Young and Evans 1978, Kuntz 1982). Perennial forbs reproduce slowly until the community becomes closed. Productivity and basal cover are then reduced as perennial grasses and shrubs dominate the site. Perennial forbs may remain in the community for a long time in a suppressed state until another fire or other disturbance occurs.

The amount of perennial forbs present in sagebrush-grass vegetation varies greatly by habitat type. For example, the Artemisia vaseyana/Festuca idahoensis ht. commonly includes the genera Balsamorhiza, Lupinus, Crepis, Senecio, and Castilleja (Hironaka et al. 1983). Habitat types of Artemisia wyomingensis have low perennial forb components in any successional stage. Consequently, the increase in perennial forbs the first 3 years following fire on these habitat types is low when compared to that of the Artemisia vaseyana communities.

The shrub component of most sagebrush-grasslands is dominated by Artemisia. Since most species do not resprout, the cover of the shrub layer is greatly reduced following fire. Establishment of seedlings varies by species. Artemisia vaseyana seedlings rapidly establish and densities the first year can equal that of the preburned condition (Kuntz 1982). If perennial plant competition is low, densities may actually be greater after the fire (Hironaka et al. 1983). Young and Evans (1978) observed only minor seedling establishment of Artemisia tridentata (probably subsp. wyomingensis) four years after a wildfire. This subspecies does not have the reproductive potential of A. vaseyana.

Growth rates of the Artemisia species varies as well as the productive potential of the sites on which they occur. The rapid growth rates of Artemisia vaseyana and high site potential of areas on which it occurs, result in cover equalling the preburn level within 15 to 25 years after the fire. The other subspecies of Artemisia tridentata may be reduced in cover and density for over 50 years following fire.

Artemisia will invade the site, regardless of the herbaceous plant competition and eventually dominate the community to the detriment of the perennial grasses (West et al. 1984). After 13 years of protection from grazing, Artemisia seedlings continued to establish and the perennial grasses declined as it increased. Only annual grasses were found to increase in density during this time.

Most, but not all, populations of Chrysothamnus resprout but the seedling establishment potential varies locally. Many subspecies of Chrysothamnus have been described. Probably much of the variation observed could be explained if we had a better understanding of the response of the subspecies. The greatest increases in density after fire are in the more arid portions of the sagebrush-grass vegetation. In these habitat types the density and cover may increase dramatically following fire, particularly after repeated fires. Chadwick and Dalke (1965) found that cover of Chrysothamnus viscidiflorus increased over 400% as a result of fire. Young and Evans (1974) found few seedlings on undisturbed sites. Populations will eventually begin to decline after 20 years, but individuals may persist in the community for 50 years.

Conclusions

The perceived successional and community changes resulting from fire in sagebrush-grasslands are largely dependent upon the descriptive parameter used. Total biomass production is usually reduced as a result of fire but there is an increase in herbaceous production. The duration of this increase varies by habitat type and ecological condition. The more rapid the re-establishment of Artemisia, the shorter the time of increased herbaceous production.

Herbaceous production of annual plants increases during the first 2 to 5 years. Production of these plants will be suppressed as the perennial plants begin to dominate the community. Communities which are depleted of herbaceous plants or areas in which the fire causes high plant mortality will often be dominated by annuals until the re-establishment of Artemisia occurs. In some habitat types this period may exceed 50 years.

Shrub production immediately after the fire is low unless there is substantial amounts of Chrysothamnus present prior to the burn. On these sites, Chrysothamnus may dominate the vegetation until the re-establishment of Artemisia. This condition is more common in more arid habitat types which are repeatedly burned within a short time period.

Basal cover of most species is reduced the first year after the fire. After this the herbaceous plants will begin to recover. An increase in density of annuals and size of surviving perennial herbaceous plants will result from the reduction of shrub competition and therefore, the basal cover of the herbaceous plants will increase. Basal cover of the herbaceous plants will then decline as the shrubs regain dominance in the community.

The effect of fire on shrub density is dependent upon the species, habitat type and condition of the site. Density of Artemisia vaseyana, A. cana or A. tripartita may remain approximately the same or even increase after the fire. Other species of Artemisia and Purshia tridentata may be reduced on the site for 15 to over 50 years. Considerable numbers of Chrysothamnus spp. or Tetradymia spp. may become established if there is a seed source and perennial competition is low. The initial establishment or lack of establishment of shrubs during the first 5 years after the fire will affect the community composition for many years.

Most herbaceous plants do not increase in density significantly during the first 3 to 5 years after burning. Rather, establishment occurs more slowly over a long time period until the community becomes closed. Large numbers of new plants can be observed during some years. This seems to be more a function of seed availability and favorable weather conditions on a given year and less a function of time since the fire occurred. The process of seedling establishment is not understood.

The importance of stratifying vegetation into classification units cannot be over stressed. Identifying fire effects by habitat type has improved the predictability of fire effects. In some instances it has been necessary to use smaller units than the habitat type such as phases or soil types. Similarly, identifying effects on plants by subspecies or varieties has improved predictability. This has occurred with Artemisia and probably would improve the predictability of Chrysothamnus.

The effect of fire on plant community descriptors such as herbaceous productivity, basal or foliar cover and density is variable. As a result, the measured effect on community composition and subsequent succession stages is variable. If biomass production is considered, distinct successional stages are evident. If plant density is used as a criterion, then for many species only change may be apparent as a result of the fire.

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Effects of Fire on Grasses and Forbs
in Sagebrush-Grass Communities

Henry A. Wright

Abstract

Effect of fire on grasses is largely determined by season of burn, size of plant, amount of dead material, growth form, species, precipitation, and whether is is an annual or perennial. Burning in June or July is the most detrimental time to burn bunchgrasses. Before or after these months, fire is less detrimental. Spring burns (April) are less detrimental than fall burns (September-October), although most species recover in 1 to 3 years. Small climax plants are more resistant to fire than large plants. Rhizomatous species such as Agropyron dasystachyum and A. smithii tolerate fire well. Seral species such as Poa sandbergii and Sitanion hystrix also tolerate fire quite well at any time of the year. Leafy bunchgrasses (Stipa, Fescue) are slower to recover than stemmy bunchgrasses (Agropyron sp., Elymus, Sitanion). Early maturing species (Poa sandbergii and Bromus tectorum) tolerate summer fires well. Annuals such as cheatgrass (Bromus tectorum) are very tolerant to fire because the seed is resistant to fire. Moreover, this species will burn when leafy perennials can be killed. Thus, frequent fires will promote cheatgrass. Fires in June can temporarily suppress medusahead (Taeniatherum asperum), but it behaves as a climax species. If no grasses are in the understory of sagebrush (Artemisia sp.), major rennovation, including seeding, will be needed to restore a community to a sagebrush-grassland.

Most forbs tolerate fire well if burned in spring or fall. Only forbs that remain green the year round such as Eriogonum sp. are severely hurt by prescribed burns. Perennial forbs are usually fully recovered at the end of the second growing season. Fire can enhance the number and diversity of forbs because they have hard seed that can be scarified by a fire.

Distribution, Climate, Soils, and Vegetation

Sagebrush-grass vegetation covers at least 39.1 million hectares in the Western United States (USDA Forest Service 1936), but probably considerably less than the 109 million hectares estimated by Beetle in 1960 (Tisdale et al. 1969). The largest contiguous area lies in eastern Oregon, southern Idaho, southwestern Wyoming, northern Utah, and northern Nevada (Vale 1975). Most of this area occurs below the pinyon-juniper zone, but in the absence of a pinyon-juniper zone, sagebrush-grass vegetation will border curlleaf mahogany (Cercocarpus ledifolius), Gambel oak Quercus gambelii), ponderosa pine (Pinus ponderosa) or Douglas-fir (Pseudotsuga menziesii). Sagebrush-grass communities also occur above the pinyon-juniper zone in the Great Basin (Billings 1951) and throughout most mountain plant communities in the Rocky Mountain and Intermountain regions (Beetle 1960).

Most sagebrush-grass is found at elevations from 610 to 2,135 m. The sagebrush-grass zone also occurs below 305 m in south central Washington and British Columbia and mixes with all vegetation zones to varying degrees up to 3,050 m (Beetle 1960), including the subalpine herbland. Where sagebrush-grass prevails below 2,135 m, annual precipitation varies between 20 to 50 cm (Tisdale et al. 1969). Soil texture varies from loamy sand to clay (Tisdale et al. 1969). Most soils are derived from basalt, although extensive areas have soils derived from rhyolite (southeastern Oregon and Nevada), loess, lacustrine, alluvium, and limestone. Interactions of soils, precipitation, and elevation results in many distinct combinations of sagebrush-grass dominated ecosystems.

Dominant grasses include bluebunch wheatgrass (Agropyron spicatum), Idaho fescue (Festuca idahoensis), needle-and-thread (Stipa comata), Thurber needlegrass (Stipa thurberiana), and to a lesser extent, Indian ricegrass (Oryzopsis hymenoides). All of these species or only one may be present in a particular understory. Needle-and-thread and Indian ricegrass dominate sandy soils throughout the sagebrush-grass zone. On other soils, bluebunch wheatgrass dominates areas with moderate annual precipitation (22 to 35 cm) and Idaho fescue dominates the most mesic sites, generally those with more than 35 cm of annual precipitation. Temperature interacts with precipitation, so the moisture threshold that separates bluebunch wheatgrass from Idaho fescue can vary from 30 to 40 cm of precipitation annually. Thurber needlegrass occurs on medium-textured soils in an 20- to 30-cm precipitation zone. Sandberg bluegrass (Poa sandbergii) and bottlebrush squirreltail (Sitanion hystrix) are the most common subdominant bunchgrasses. Junegrass (Koeleria cristata) and other Poa spp. are also often present if the annual precipitation is above 28 cm. Rhizomatous grasses that occupy localized areas include thickspike wheatgrass (Agropyron dasystachyum), plains reedgrass (Calamagrostis montanensis) and riparian wheatgrass (Agropyron riparium). Cheatgrass (Bromus tectorum), an introduced annual, occupies millions of acres on disturbed ranges (Klemmedson and Smith 1964). Medusahead (Taeniatherum asperum), another introduced annual, occupies disturbed clay sites that have well-developed profiles (Dahl 1966).

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Forbs are present in great variety and abundance in climax communities where the precipitation is in excess of 28 to 30 cm per year. They may account for as much as 50% of the herbaceous production in eastern Idaho. For this reason herbicides, at least in our opinion, are undesirable to manage sagebrush-grass communities in eastern Idaho where balsamroot (Balsamorhiza sagittata) and lupine (Lupinus sp.) are typically the most abundant forbs. Forbs account for only 5 to 15% of the herbaceous vegetation in eastern Oregon. Groundsel (Senecio sp.), tapertip hawksbeard (Crepis acuminata), western yarrow (Achillea millefolium), and locoweed (Astragalus sp.) are the most common forbs and recover within 3 to 4 years after the use of herbicides (Britton and Sneva 1981).

Fire History

Before the influence of man, fire covered contiguous units of sagebrush-grass communities in northern Yellowstone National Park at an average frequency of 32 to 70 years (Houston 1973). Within a large portion of this locale, however, fire swept smaller areas at least every 17 to 41 years. Dating all fires that occurred within a locale, Houston theorized that the frequency of fire in sagebrush communities within Yellowstone Park was 20 to 25 years. This estimated frequency was based on the assumption that the record of fire scars for any one tree underestimated the frequency of fire because not all trees were scarred by every fire. Houston's findings that many but not all trees had fire scars with similar dates suggests that once burned, an area was unlikely to have sufficient fuel to reburn for several years.

Based on the vigorous response of horsebrush (Tetradymia canescens) to fire and the 30-plus years that are needed for it to decline to a low level after a fire (Harniss and Murray 1973), probable frequency of fire in eastern Idaho would be about 50 years. If fires occurred every 20 to 25 years, as Houston implies, many sagebrush-grass communities could be dominated by horsebrush and rabbitbrush. In Nevada intervals between fires in big sagebrush communities may be as long as 90 years (Young and Evans 1981).

Ecological Effects of Fire

Grasses and Forbs

The effects of fire on grasses depends largely on growth form and season of burning. Bunchgrasses with densely clustered culms, such as Idaho fescue and needle-and-thread, can be severely harmed by fire (Blaisdell 1953; Wright 1971), especially if burned during June or July (Wright 1971). Their dense culms will burn 2 to 3 hours after a fire passes. Temperatures as high as 538°C will be reached 45 min after a fire has passed (Wright 1971). Thus, many plants often die or have only a few culms that survive, regardless of the intensity of the passing fire. Late summer and fall burns are the least harmful. Often forage yields in seeded crested wheatgrass (Agropyron desertorum) stands will increase 3 to 6-fold three years after a burning treatment because of reduced competition from big sagebrush (Ralph and Busby 1979).

Threadleaf sedge (Carex filifolia) also has a compact growth form and is severely harmed by fire (Vallentine 1971). Idaho fescue is very sensitive to summer and fall fires where the precipitation is marginal for its existence (Blaisdell 1953; Conrad and Poulton 1966). Research in eastern Oregon indicates that Idaho fescue will recover in 2 years if burned during October (Britton and Sneva 1981; Britton et al. 1983).

Bluebunch wheatgrass, bottlebrush squirreltail, and the crested wheatgrasses (Agropyron cristatum, A. desertorum, and A. sibericum) are less susceptible to fire injury than Idaho fescue or Stipa sp. (Blaisdell 1953; Conrad and Poulton 1966; Wright 1971; Vallentine 1971) because the former are mostly coarse stems with a minimum of leafy material. They burn quickly and little heat transfers below the soil surface (Wright 1971). Moreover, the small size of Sandberg bluegrass and bottlebrush squirreltail in climax communities helps them survive fires (Wright and Klemmedson 1965); therefore they usually increase in abundance after a fire. All rhizomatous grasses such as thickspike wheatgrass and plains reedgrass increase immediately after a fire (Blaisdell 1953). Production from rhizomatous grasses on burned plots will be above that on controls for about 30 years (Harniss and Murray 1973).

Bluebunch wheatgrass will return to preburn production in 1 to 3 years (Blaisdell 1953; Moomaw 1957; Conrad and Poulton 1966; Uresk et al. 1976; Daubenmire 1963, unpublished progress report); needle-and-thread in 3 to 8 years, depending on site (Blaisdell 1953; Dix 1960; Wright 1977, unpublished observation); Thurber needlegrass (Stipa thurberiana) in more than 3 years (Uresk et al. 1980); and Idaho fescue in 2 to 12 or more years, depending on soil moisture, season, and intensity of the fire (Blaisdell 1953; Conrad and Poulton 1966; Harniss and Murray 1973; Britton and Sneva 1981). The response of prairie junegrass (Koeleria cristata) to fire is similar to that of needle-and-thread (Vallentine 1971). Cusick bluegrass (Poa cusickii) is reduced 50% the first growing season after burning (Uresk et al. 1976) and may remain reduced for 3 years after an August wildfire (Uresk et al. 1980). Indian ricegrass (Oryzopsis hymenoides) is only slightly damaged by fire (Vallentine 1971).

Repeated burning every few years or burning in early summer will deplete a stand of perennial grasses and allow annual grasses, chiefly, cheatgrass (Bromus tectorum), to increase sharply (Pickford 1932; Wright and Klemmedson

1965). Once a sagebrush-grass community is depleted of perennial plant cover, secondary succession goes from Russian thistle (Salsola kali) to mustard (Sisymbrium and Descurainia sp.) to cheatgrass within 5 years (Piemeisel 1951). Pechanec and Hull (1945) found that burning near Boise, Idaho, reduced cheatgrass plants in varying numbers, depending on the month of the burn, as follows:

Burn	Cheatgrass plants	
(Month)	(ft^2)	(m^2)
June	14	15
July	11	12
August	41	44
October	45	48
November	124	133

The early summer burns are only a temporary setback for cheatgrass at a time of the year when climax perennials are easily killed by fire (Wright and Klemmedson 1965). Hence, the density of cheatgrass increases over time while fewer perennials survive after each fire. Excluding livestock from cheatgrass ranges or sites with dense brush for 13 years will not improve the quantity or composition of native forages (West et al. 1984).

Cheatgrass areas can only be reclaimed by chemical fallow techniques (Eckert and Evans 1967) or plowing and then seeding. Most seeding has been done with wheatgrass (Hull 1971). Fairway wheatgrass (Agropyron cristatum), crested wheatgrass (A. desertorum), and Siberian wheatgrass (A. sibericum) are well adapted to this zone. Fairway wheatgrass is best adapted to moderately mesic sites and Siberian wheatgrass is best adapted to the driest sites in the 20- to 30-cm precipitation zone. Seeding dense sagebrush areas after fire can be successful with good grazing management techniques (Evans and Young 1978). Otherwise, such seedings are failures.

Fall burning does not harm most forbs because many of them are dry and often disintegrated by this time. However, some forbs remain green and are very susceptible to fire. Pechanec et al. (1954) classified forbs according to their susceptibility to fire.

After 12 years, Blaisdell (1953) found that only the heavy sagebrush-grass burn (all sagebrush plants consumed by fire) supported more forbs than the control 12 years after burning. By the end of 30 years, forbs had returned to preburn levels (Harniss and Murray 1973), although both burned and unburned plots contained at least 5 times as many forbs as before the burn.

In a sagebrush-bunchgrass community in British Columbia, Johnson and Strang (1983) found that a July 26, 1979, wildlife enhanced many forbs the year after a burn. Antennaria Microphylla was especially enhanced by fire.

Spring burns have been limited in sagebrush grass communities and little data has been reported. However, an April burn on the U.S. Sheep Experiment Station in eastern Idaho showed that grass yields of Idaho fescue and thickspike wheatgrass were not harmed at the end of the first growing season (Blaisdell et al. 1982). Forbs were reduced 43% during the first growing season, but were 181% of the unburned by the second growing season. These results are comparable to my observations of spring burns in western Wyoming, central Idaho, and Nevada. Thus, where possible, spring burning should be encouraged in sagebrush-grass communities.

Fire Effects Data For Plant Species

Response of Grasses to Burning

Bluegrasses

In general, bluegrasses (Poa sp.) are slightly damaged by burning. Wright and Klemmedson (1965) observed no change in basal area of Sandberg bluegrass (Poa sandbergii) during any season regardless of the size of

Table 1. Susceptibility of forbs to fire by three damaged classifications at Dubois, Idaho (Pechanec and others 1954)

Severely damaged	Slightly damaged	Undamaged
Antennaria dimorpha	Astragalus sp.	Achillea lanulosa
Antennaria microphylla	Castilleja angustifolia	Allium sp.
Arenaria uintahensis	Crepis acuminata	Arnica fulgens
Erigeron engelmannii	Geranium viscocissimum	Balsamorhiza sagittata
Eriogonum caespitosum	Lupinus caudatus	Comandra umbellata
Eriogonum heracleoides	Penstemon radicosus	Erigeron corymbosus
Phlox canescens	Sphaeralcea munroana	Lupinus leucophyllus
		Phlox longifolia
		Senecio integerrimus
		Sisymbrium linifolium
		Zygadenus paniculatus

plants. These plants were mature, dry, and varied in diameter from 2.5 to 7.5 cm. Tisdale (1959) reported some damage to Sandberg bluegrass in communities with 7 to 14% sagebrush cover. This damage was possibly caused by plants being old and pedestaled. High mortality has been observed in southern Oregon when plants are pedestaled (Hammersmark 1977, personal communication). An August wildfire in northeastern California caused decreases in plant numbers (Countryman and Cornelius 1957). Moomaw (1957) found no damage in eastern Washington. Uresk et al. (1980) measured a decrease in basal area of cusick bluegrass (P. cusickii) 3 years after an August wildfire.

On the upper Snake River Plains of Idaho, Nevada bluegrass (P. nevadensis) and Sandberg bluegrass showed little change in production for 3 years after burning (Harniss and Murray 1973). This initial static period was followed by increased yields, with the burned area producing about 1.5 times more than the unburned. Thirty years after burning, yield was substantially lower on both burned and unburned areas, although the burned area was producing twice as much as the unburned. Big bluegrass (P. ampla) is not mentioned in the literature but, due to larger clone size and greater potential accumulation of litter in the crown, it would be expected to incur slightly more damage than other bluegrasses.

Cheatgrass

Cheatgrass (Bromus tectorum) is not appreciably affected by burning although production may be reduced for the first year. Abandoned fields on the Snake River Plains dominated by cheatgrass were changed to primarily tumblemustard (Sisymbrium altissimum) and Russian thistle (Salsola kali) after burning for 2 to 3 years. Cheatgrass dominated these fields during the next 2 to 3 years (Piemeisel 1938). Burning was found to reduce stands of cheatgrass in eastern Washington, presumably because of seed destruction (Robocker et al. 1965). Depending on the intensity of burn, germinable cheatgrass seed can be reduced 80 to 99% (Young et al. 1976). This reduction left from 32 to 374 germinable seeds/m^2, but as few as 54 cheatgrass seeds/m^2 moderately reduced establishment of crested wheatgrass. June and July burns reduced cheatgrass plant numbers to 150 and 118/m^2 compared to 439, 481, and 1,327 plants/m^2 on August, October, and November burns, respectively, near Boise, Idaho (Pechanec and Hull 1945). Density reductions are only temporary, for surviving plants are 25 times larger than those on control and produce abundant seed the year after a burn. The second year after a burn seed densities are comparable to unburned plots (Young and Miller 1983). Early summer burns will kill perennial grasses and allow cheatgrass to increase sharply.

Cheatgrass can rapidly occupy a burned area if only a few seeds are available (Countryman and Cornelius 1957; Young and Evans 1978; Merrill et al. 1980). Barney and Frischknecht (1974) reported that cheatgrass cover declined during the first 22 years after fire, then stablized. This cover change varied from 12.6% on 3-year-old burns to 0.9% on the oldest burns.

Idaho Fescue

The majority of evidence indicates that Idaho fescue (Festuca idahoensis) is severely damaged by summer and fall burns (Pechanec and Stewart 1944; Blaisdell 1953; Harniss and Murray 1973). After a period of 30 years, Idaho fescue was just approaching its former abundance on the upper Snake River Plains (Harniss and Murray 1973). However, the annual precipitation is 35 cm in this area, which is marginal for Idaho fescue in this region of Idaho. As a result of a summer wildfire in eastern Washington, Idaho fescue mortality was 27%, with a reduction in basal area of 50% (Conrad and Poulton 1966). In northeastern California, basal area of Idaho fescue was reduced approximately 80% by an August wildfire (Countryman and Cornelius 1957).

Mid-May burns in eastern Oregon resulted in 30% mortality and a 48% reduction in basal area (Britton et al. 1983). However, when the plants were dormant in the fall, no mortality resulted although there was a 34% reduction in basal area. Phillips (1977, personal communication) observed that in central Oregon, wildfires were more damaging to Idaho fescue on coarse soils than on fine-textured. soils. Good soil moisture was found beneficial to Idaho fescue survival during spring burns in Nevada (Beardall and Sylvester 1976). More recent work by Blaisdell et al. (1982) showed that Idaho fescue was not harmed at all by spring burning when harvested at the end of the first growing season.

Indian Ricegrass

Indian ricegrass (Oryzopsis hymenoides) is important in sagebrush-bunchgrass communities only in localized situations. As such, it has not been the subject of intensive investigations. Pechanec and Stewart (1944) mention it as being slightly damaged and slow to increase after burning.

In west-central Utah, Indian ricegrass was found to be an important species on burned areas (Barney and Frischknecht 1974). Therefore, it probably has good survival characteristics. Spring burning in Utah did little damage to Indian ricegrass; growth began about 3 weeks after burning (Jensen 1977, personal communication). Summer wildfires in Nevada reduce basal area, but little mortality was noted (Wagner 1977, personal communication).

Junegrass

In eastern Oregon, junegrass (Koeleria cristata) has been found to be one of the most fire-resistant perennial bunchgrasses (Britton et al. 1983). Burning in mid-May reduced basal area by 32%, with 20% mortality. Burning in mid-June just after seed-set reduced basal area 18%, with no mortality; burning in mid-October produced only slightly more damage. Light damage is probably due to the relative small size of the typical junegrass clone.

Fall burning in North Dakota increased the frequency of junegrass on a sandy soil (Dix 1960). Twelve years after burning in Idaho, junegrass yield was higher on burned areas than on unburned areas (Blaisdell 1953). Countryman and Cornelius (1957) reported a slight decrease in junegrass due to a wildfire although the sample was too small for adequate interpretation. Similarly, Koeleria micrantha was not harmed by a July wildfire in British Columbia (Johnson and Strang 1983).

Needlegrasses

Most needlegrasses (Stipa sp.) are damaged by burning, especially during the first year. Harniss and Murray (1973) reported a severe reduction the first year after burning needle-and-thread (Stipa comata). Season of burn rather than burning intensity or plant size was found to be the most critical factor in mortality of needle-and-thread (Wright and Klemmedson 1965). June burns killed all of the small plants and 90% of the large plants. In July, 20% of the burned plants died but no mortality was recorded for August treatments. Among large plants, the average basal area reduction after June burns was 99.6%; July burns, 96%; and August burns, 68%. The reduction in basal area for the small plants following June burns was 100%, July burns, 82%. Small plants burned in August exhibited some thinning of the crown. This damage was related to the intolerance of needle-and-thread grass to herbage removal and the large amounts of dead material per unit basal area (Wright 1971). In western North Dakota, fall burning decreased needle-and-thread grass frequency by 11% on sandy soils but increased frequency by 10% on a clayloam soil. Observations in southern Idaho indicate that with moderate grazing treatments, needle-and-thread grass requires 4 to 8 years after burning to fully recover.

Twelve years after burning, Blaisdell (1953) observed that needle-and-thread grass and Columbia needlegrass (S. columbiana) were not significantly affected by any intensity of burn, although the former produced 11 to 29 kg/ha on burned than on unburned range. Western needlegrass (S. occidentalis) was reduced the first year after an August wildfire in northeastern California (Countryman and Cornelius 1957). By the third year after burning, western needlegrass had almost doubled in basal area as compared to the unburned area.

Thurber needlegrass (S. thurberiana) is probably the least fire resistant needlegrass. Uresk et al. (1980) found that an August wildfire reduced the basal area for at least 3 years, with a concurrent decrease in leaf length. In eastern Oregon, Thurber needlegrass was severely damaged by burning (Britton et al. 1983). Plants burned in mid-May had 80% mortality and the basal area was reduced by 93%. In mid-June, mortality increased to 90% and basal area was reduced 93%. Least damage resulted from October burns, with no mortality and a 48% reduction in basal area. Wright and Klemmedson (1965) reported similar results for Thurber needlegrass.

Giant wildrye and Great Basin wildrye

Wildrye is usually restricted to flood plains or basin bottoms that are saline and/or alkaline (Lesperance et al. 1978). Based on my observations of this plant in eastern Oregon and Nevada it is very tolerant to fire and yields will increase several-fold when associated shrubs have been suppressed.

Sedges

Sedges show various responses to burning. Pechanec and Stewart (1944) list threadleaf sedge (Carex filifolia) as being severely damaged; Douglas sedge (C. douglasii) was classed as undamaged. This difference was attributed to the ability of Douglas sedge to initiate growth from basal buds. Twelve years after burning, Blaisdell (1953) reported sedges were producing more on light burns but less on moderate and heavy burns in one area while the opposite trend was found on another area. Threadleaf sedge was found to be producing on the average more on burned areas as compared to unburned areas, therefore initial damage by burning was not permanent. Douglas sedge was reduced in number of plants as a result of a wildfire in northeastern California (Countryman and Cornelius 1957).

Bottlebrush Squirreltail

Bottlebrush squirreltail (Sitanion hystrix) is one of the more fire resistant bunchgrasses, although some damage is apparent. In southern Idaho, Wright and Klemmedson (1965) observed no plant mortality as a result of burning. There was some reduction in basal area for plants burned in June but it was most apparent (15% reduction) in July for both large and small plants. In August only the large plants responded to hot burns, with a 16% reduction in basal area. The greater overall damage in July was probably due to higher initial burn temperatures in the plant crown. Wright (1971) reported that burning generally reduced herbage production of bottlebrush squirreltail most during May and somewhat less thereafter. In northeastern California, bottlebrush squirreltail plants were found to increase in winter the first year after wildfire but decreased by the third year (Countryman and Cornelius 1957).

Burning bottlebrush squirreltail plants in a drought year in eastern Oregon resulted in 30% mortality in mid-May (Britton and Sneva 1981). No mortality was recorded for mid-June or October burns. The mid-May burns reduced basal area by 73% while the October burn reductions were 48%.

In west-central Utah, bottlebrush squirreltail cover was found to increase during the first 5 to 6 years after burning (Barney and Frischknecht 1974). This increase was stable for up to 40 years. Often bottlebrush squirreltail plants are very small and will increase in size rapidly after a burn. The larger plants, however, would be slightly harmed (Wright 1971; Britton and Sneva 1981).

Wheatgrasses

Fall burning of crested wheatgrass (Agropyron desertorum) results in only small changes in the stand. Density of plants should remain unchanged (Kay 1960), although yield may be reduced during the first growing season after burning (Lodge 1960). In a decadent stand of crested wheatgrass that had been invaded by sagebrush, Ralph and Busby (1979) found that crested wheatgrass yields increased from 112 kg/ha to 703 kg/ha over a 3-year period after a burn. It was rested from livestock for those 3 years. Crested wheatgrass seedings are considered fire resistant because many observers report that wildfires move only 2 or 3 m into a seeding.

The negative effects of burning bluebunch wheatgrass are usually evident only in the first year after burning. Uresk et al. (1980) measured decreases in leaf lengths and basal area, but an increase in yield 3 years after burning in eastern Washington. These results are similar to the 29% reduction in basal area and 1% mortality observed in the same region by Conrad and Poulton (1966). In eastern Oregon during mid-May, burning decreased the basal area by 78% with a 50% mortality. When plants were burned during the fall, there was no mortality but a reduction in basal area of 47% compared to preburn measurements of the same plants (Britton and Sneva 1981). Effects of burning (Wright 1974) were magnified by an extremely dry year. Bluebunch wheatgrass will usually return to preburn production in 1 to 3 years (Blaisdell 1953; Moomaw 1957; Conrad and Poulton 1966; Uresk et al. 1976; Johnson and Strang 1983).

Bluebunch wheatgrass (A. spicatum) is slightly affected by burning. Twelve years after burning, Blaisdell (1953) found an almost two-fold increase in yield compared to unburned controls. After 30 years, yield of bluebunch wheatgrass on the same area was slightly below the controls (Harniss and Murray 1973). Cover of bluebunch wheatgrass remained uniform in west-central Utah for 40 years after burning before the juniper overstory caused a decline (Barney and Frischknecht 1974).

Response of other wheatgrasses falls somewhere between crested wheatgrass and bluebunch wheatgrass with the exception of the rhizomatous wheatgrasses. Thickspike wheatgrass (A. dasystachyum) exhibits virtually no change one year after burning and after 12 years produces about twice as much as unburned controls (Blaisdell 1953). It appeared that the more intense the burn the greater the response. In northeastern California, a mixed wheatgrass stand was burned in the fall of the third growing season (Kay 1960). The following summer, a 25% increase in stocking was measured, primarily due to rhizomes of intermediate wheatgrass (A. intermedium) and Pubescent wheatgrass (A. trichophorum). Tall wheatgrass (A. elongatum) remained unchanged. In North Dakota, western wheatgrass (A. smithii) was unchanged in frequency regardless of site or soil (Dix 1960).

Response of Forbs to Burning

Forbs generally respond better to burning than grasses when burned in late summer and fall. Where plant communities have large proportions of forbs in the herbaceous component, burning may provide the best manipulation technique. Fall burning does not harm most forbs (Pechanec and Stewart 1944; Young and Evans 1978; Merrill et al. 1980; Johnson and Strang 1983) because they are often dry and disintegrated by this time. Pechanec and Stewart (1944) classified forbs according to their susceptibility to fire (Table 1).

Due to lack of research evidence on individual forb species, no attempt will be made to review each species mentioned in the literature. Instead, each article mentioning forbs will be abstracted.

Probably the best research treatment of forbs is from the prescribed burns conducted in Clark and Fremont Counties on the upper Snake River Plains of Idaho. This work is presented in a series of three articles by Pechanec et al. (1954); Blaisdell (1953); and Harniss and Murray (1973). Pechanec et al. (1954) observed that the rapidity of increase by the lightly damaged or undamaged species depended largely on whether the plant spreads by rootstocks. Those that do not, even though undamaged, increase slowly after burning. These include some of the more palatable species such as arrowleaf balsamroot (Balsamorhiza sagittata) and tailcup lupine (Lupinus caudatus). Recovery rates of balsamroot and lupine in this study are supported by Young and Evans (1978) and Merrill et al. (1980).

Despite a quick recovery from burning, an increase in number of plants must await seed production.

Plant species spreading by rootstocks or root shoots are least harmed and spread most rapidly after burning. These species include western yarrow (Achillea lanulosa), purpledaisy fleabane (Erigeron corymbosus), longleaf phlox, (Phlox longifolia), flaxleaf plainsmustard (Sisymbrium linifolium), lambstongue groundsel (Senecio integerrimus), orange arnica (Arnica fulgens), and common comandra (Comandra umbellata). Such species as western yarrow, longleaf phlox, and purpledaisy fleabane doubled in production within 3 to 4 years.

Approximately 12 years after the burns in Clark and Fremont Counties, Blaisdell (1953) reevaluated the vegetation response. In Fremont County, he found total forb production was considerably higher on all burn intensities as compared to unburned areas. Forb production in kilograms per hectare was unburned, 145; light burn, 215; moderate burn, 265; and heavy burn, 191. Of the species mainly responsible for the higher yield of forbs on burned areas, western yarrow, aster (Aster sp.), fleabane, and goldenrods (Solidago sp.) are rhizomatous perennials. Littleleaf pussytoes (Antennaria microphylla), a suffrutescent forb of low forage value, and stick geranium (Geranium viscosissimum), a perennial rated fair forage, also contributed to the higher yield, especially on light and moderate burns. On the other hand, yield of knotweed (Polygonum douglasii), an undesirable annual, was greatest on the heavy burn. The lower yield of plumeweed (Cordylanthus ramosus) on burned areas compared to unburned areas approached statistical significance.

The 1934 inventories showed a marked increase in total production of forbs on burned areas in relation to that on the unburned. This trend continued through the third year, especially on burns of light and moderate intensity. Although much of these early increases had disappeared by 1948, differences were still significant for light and moderate burns.

Rhizomatous species on all burn intensities showed relative increases the first year after burning, but subsequent trends were variable. On the other hand, suffrutescent forbs [pussytoes and eriogonum (Eriogonum caespitosum and E. heracleoides)], decreased markedly, roughly proportionate to burn intensity, and then increased. Annuals, primarily gayophytum (Gayophytum diffusum), knotweed, plumeweed, and goosefoot (Chenopodium sp.), made enormous relative increases in 1934, roughly in proportion to burn intensity. Portions of these relative increases persisted through 1936, but had disappeared by 1948 on all but the heavy burn. The persistence of annuals on the heavy burn was shown by actual yield of knotweed, 29 kg/ha on the heavy burn as compared to 8 kg/ha on the control. Other perennial forbs generally showed an initial but temporary increase after burning.

After 12 years, only the heavy burn in Clark County supported a significantly higher yield of forbs compared to the unburned area. It appeared that fleabane and phlox (both rhizomatous species) on burns of all intensities and lupines on the heavy burn were producing more than on the unburned ground. Apparently the effect of burning on the other forbs was negligible after 12 years.

In contrast with Fremont County, inventories of Clark County plots the year after burning showed a decrease in total forb production on burned areas in relation to the unburned. By the third year, considerable increases in relative yield were evident, but most of these early effects disappeared during the next 9 years. As in Fremont County, rhizomatous forbs generally increased the first year, but suffrutescent species, eriogonum and pussytoes, decreased markedly on all burns. Rhizomatous species continued to increase through the third year, then decreased. After the initial relative decreases, suffrutescent species increased throughout the study period and regained much of their original losses. With the exception of plumeweed, annuals were present only in very small amounts. Other perennial forbs increased the first year on burns of all intensities, but trends in following years were not well defined.

By 1966, Harniss and Murray (1973) found little difference in forb production on burned and unburned plots. Although individual species were not mentioned, perennial forbs accounted for the bulk of the production.

Abandoned fields on the Snake River Plains dominated by cheatgrass were burned during the early 1930's (Piemeisel 1938). The first year after burning the major portion of the vegetation was tumblemustard and Russian thistle, with some flixweed tansymustard (Descurainia sophia). It took 2 to 3 years for the fields to again be dominated by cheatgrass. Near Dubois, Idaho, Mueggler and Blaisdell (1958) reported an increase in forb production after burning. Those species most benefited included timber poisonvetch (Astragalus convallarius), purpledaisy fleabane, and lupine. Forbs that were injured include littleleaf pussytoes and matroot penstemon (Penstemon radiocosus).

In western North Dakota, Dix (1960) found that wild lettuce (Lactuca pulchella) decreased about 20%. The most dramatic decrease (63%) was on loamy fine sand. Red globemallow (Sphaeralcea coccinea) increased when present in the vegetation.

Robocker et al. (1965) found that burning decreased most forbs on a sandy soil in eastern Washington. Burning appeared to have reduced stands of tumblemustard, tansymustard (Descurainia pinnata), and whitlow-wart (Draba verna).

In west-central Utah, Barney and Frischknecht (1974) sampled burns ranging in age from 3 to more than 100 years. The most abundant forbs during the first stages of succession were pale alyssum (Alyssum alyssoides), flixweed tansymustard, sunflower (Helianthus annuus) coyote tobacco (Nicotiana attenuata), and Russian thistle. Because these forbs were abundant on recent burns, it can be assumed that they were not seriously damaged by burning.

Yarrow was found to decrease the first 2 years after burning in northeastern California (Countryman and Cornelius 1957). However, by the fifth year, there was a 7.5-fold increase in crown area.

In eastern Oregon, tailcup lupine changed slightly in cover from an average of 15% to more than 16% 1 year after fall burning (Britton et al. 1983). The burns were in a dry year followed by a drier year. Hammersmark (1977, personal communication) observed in southern Oregon that astragalus, arrowleaf balsamroot, tapertip hawksbeard (Crepis acuminata), tailcup lupine, globemallow, and foothill deathcamas (Zigadenus paniculatus) were not damaged by wildfire.

In Nevada, Wagner (1977, personal communication) measured a three-fold increase in frequency of longleaf phlox with no reduction in wild onion, astragalus, tapertip hawksbeard, lupines, globemallow, and foothill deathcamas as a result of wildfire. Similarly Merrill et al. (1980) found that Cerastium arvense increased after a summer burn, but most other forbs did not change in cover.

Following a July wildfire in British Columbia, Johnson and Strang (1983) found many forbs on their burned plots that were not present or unburned plots in a big sagebrush/bunchgrass community. Antennaria microphylla was the only forb that increased significantly, but Achillea millefolium, Agoseris glauca, Artemisia ludoviciana, Gaillardia aristata, Lappula echinata, Sysmbrium altissimum, Taraxacum officinale, and Trogopogon dubius were all present on burned plots compared to none on the unburned plots.

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Effects of Fire on Sagebrush and Bitterbrush[1]

Carlton M. Britton and Robert G. Clark

Community Description

The sagebrush-bunchgrass region of the Great Basin covers at least 39 million hectares (USDA Forest Service 1936). The largest contiguous area lies in eastern Oregon, southern Idaho, southwestern Wyoming, northern Utah, and northern Nevada (Vale 1975). Most of this vegetative type occurs below the pinyon-juniper zone, but in the absence of a pinyon-juniper zone, sagebrush-bunchgrass vegetation will border curlleaf mahogany (Cercocarpus ledifolius), Gambel oak (Quercus gambelii), ponderosa pine (Pinus ponderosa), or Douglas-fir (Pseudotsuga menziesii). Sagebrush-bunchgrass communities also can occur above the pinyon-juniper zone in the Great Basin (Billings 1951) and throughout most mountain plant communities in the Rocky Mountain and Intermountain regions (Beetle 1960).

Sagebrush-bunchgrass vegetation is found at elevations from 610 to 2,130 m. This zone can occur below 300 m in south-central Washington and British Columbia and mixes with all vegetation zones to varying degrees up to 3,000 m (Beetle 1960), including the subalpine herbland. Where a sagebrush-bunchgrass prevails below 2,100 m, annual precipitation varies from 20 to 50 cm (Tisdale et al. 1969). Soil texture varies from loamy sand to clay. Most soils are derived from basalt, although extensive areas have soils derived from rhyolite, loess, lacustrine, alluvium, and limestone (Tisdale et al. 1969). Interactions of soil, precipitation, and elevation result in many distinct combinations of sagebrush-bunchgrass dominated communities.

Three subspecies of big sagebrush dominate the sagebrush-bunchgrass region. These are basin big sagebrush (Artemesia tridentata ssp. tridentata), Wyoming big sagebrush (A.t. ssp. wyomingensis), and mountain big sagebrush (A.t. ssp. vaseyana). Basin big sagebrush (1 to 5 m tall) and Wyoming big sagebrush (0.4 to 0.8 m tall) are the dominants from 610 to 2130 m elevations, with the latter being the most drought tolerant (McArthur et al. 1974). Basin big sagebrush occupies a 25 to 40 cm precipitation zone on deep well-drained alluvial soils. Wyoming big sagebrush occupies a 20 to 30 cm precipitation zone on shallow soils (Tisdale et al. 1969). Mountain big sagebrush (0.75 to 1.25 m tall) is the most mesic subspecies and can be found at elevations from 1500 to 3050 m where precipitation varies from 35 to 50 cm per year (McArthur et al. 1974).

Other species of sagebrush in decreasing order of economic importance are low sagebrush (A. arbuscula), three-tip sagebrush (A. tripartita), black sagebrush (A. nova), silver sagebrush (A. cana ssp. viscidula and ssp. bolanderii), alkali sagebrush (A. longiloba), Bigelow sagebrush (A. bigelovii), and scabland sagebrush (A. rigida) (Tisdale et al. 1969, McArthur et al. 1974). The first three species generally grow below 1830 m elevation although they can occur at higher elevations. Low sagebrush occurs on shallow soils or soils with a restrictive B horizon, mostly in southern Idaho, Nevada, southeastern Oregon, and northeastern California (Fosberg and Hironaka 1964). Three-tip sagebrush occurs east of this region on mesic or dry soils in a precipitation zone of 25 to 40 cm. Black sagebrush is usually associated with calcareous soils on dry sites, but can occur on mesic sites of the Douglas-fir zone in eastern Idaho. Silver sagebrush occurs primarily in spring-flooded bottomlands and at high elevations where snow drifts. All species except three-tip sagebrush and silver sagebrush are nonsprouters. Three-tip sagebrush is a weak sprouter, and silver sagebrush is a strong sprouter.

Major shrubs associated with big sagebrush include horsebrush (Tetradymia spp.), rabbitbrush (Chrysothamnus spp.), and broom snakeweed (Xanthocephalum sarothrae). Spiny hopsage (Grayia spinosa) and mormon tea (Ephedra nevadensis) are sporadically present in the lower rainfall areas (Wright et al. 1979).

Antelope bitterbrush (Purshia tridentata) is one of the most widely distributed western shrubs. Hormay (1943) estimated its range at 138 million ha in 11 western states and southern British Columbia. Antelope bitterbrush is found from northern Arizona and New Mexico northward to southern British Columbia, and from the Cascade-Sierra Mountain Range eastward to western Montana, Wyoming, and Colorado. It occurs at elevations from 305 m in British Columbia to 3505 m in California (Hormay 1943; Nord 1965).

Bitterbrush can grow in pure stands (Stanton 1959) but more commonly in association with various genera of trees, shrubs, forbs, and grasses. For example, Franklin Dyrness (1969) described 18 habitat types in Oregon in which bitterbrush was a major component.

Bitterbrush grows on a wide variety of soils. In California, Nord (1965) reported stands on soils developed from granitic, basaltic, rhyolitic, or pumiceous parent

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materials, or on sedimentary sandstone and shale rock. In general, bitterbrush is most frequently found on young, deep to very deep, coarse-textured, well-drained soils (Driscoll 1964, Nord 1965). Bitterbrush is also a pioneer species on recent volcanic deposits in Idaho (Eggler 1941) and California (Nord 1965) and codominant with ponderosa pine on a 27-year-old mud slide in California (Dickson and Crocker 1953). Nord (1963) reported that bitterbrush often invades disturbed areas long before other plants appear and for many years may provide the only form of soil cover. Therefore, the case could be made that bitterbrush is a seral species. Management of bitterbrush as a climax species could hasten its demise in a community.

Fire Effects on Sagebrush

Big sagebrush is killed easily by fire (Blaisdell 1953). It is relatively unimportant how fast the fire moves, how hot the fire is, or what the fire intensity is. The only critical parameter is that foliage be exposed to a minimum temperature of 90°C for a period of at least 30 sec. Therefore, the effect of fire can be briefly stated: if a fire front passes through an area, the sagebrush will be killed.

A major point of concern is the length of time an area will stay free of sagebrush. On the upper Snake River Plains of Idaho, Blaisdell (1953) found that after 12 years there was only a 10% return of sagebrush after fire. However, after 30 years had passed, Harniss and Murray (1973) found sagebrush had returned to the preburn condition. Many factors influence the rate of sagebrush reinfestation on treated areas. If a seed reservoir is available, the most important factor is the pattern and amount of precipitation following the burn. If precipitation is not conducive to germination and establishment of sagebrush, the burned area may be relatively free of sagebrush for many years.

One problem facing range managers is picking a suitable area to burn. Although there are many aspects to consider prior to applying a fire treatment, one must select an area that will burn. The proper combination and amount of fuel must be present or the area will be extremely difficult to burn. An area with 7 to 10% canopy cover of sagebrush is difficult to burn and will not result in a significant increase in herbaceous yield. Figure 1 presents a fuel model that allows a manager to determine if a suitable amount of sagebrush and fine fuel are present to support a safe prescribed fire (Britton et al. 1981). The curve represents the relationship between sagebrush canopy cover and herbaceous fuel at which safe and successful prescribed burns can be expected. This relationship will hold when wind is 12 to 24 km/hr, relative humidity is 15 to 20%, and air temperature is 20 to 27 C. If burns are conducted with stronger winds and higher air temperatures at lower humidities, the curve will shift to the left. The curve will shift to the right when burns are conducted with lower winds and air temperatures in conjunction with higher humidities. As a general rule, at least 20% canopy cover of sgebrush and 300 kg/ha of herbaceous fuel is needed to ensure a successful burn.

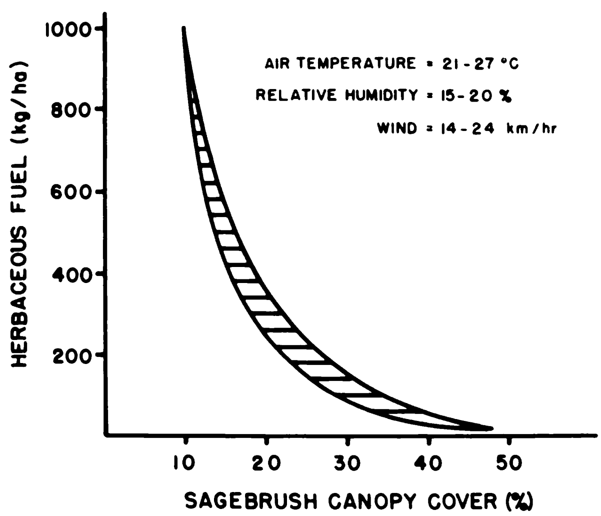

Figure 1. Fuel model of sagebrush canopy cover and herbaceous fuel load. Curve represents proportions of the two fuels where successful burns can be expected for the given conditions.

Canopy cover of big sagebrush and herbaceous fuel is greater on more productive sites. Therefore, subspecies of sagebrush can be used as an initial evaluation of site suitability. Mountain big sagebrush communities are burned most easily. Basin big sagebrush is intermediate and Wyoming big sagebrush is most difficult to burn. These differences are not related to any specific differences in individual plants but rather to sites where the subspecies occur. Mountain big sagebrush and basin big sagebrush typically occupy deeper soils on sites that receive more moisture than Wyoming big sagebrush. Thus, the better sites are capable of supporting greater plant densities. This results in more sagebrush canopy cover and herbaceous fuel. In sagebrush-bunchgrass communities, the more fuel available, the easier it is to conduct safe and effective burns.

Effects of Fire on Bitterbrush

Literature on bitterbrush has recently been summarized (Clark and Britton 1979). In this summary, 135 articles published during the period 1967-1978 were annotated, and 233 articles published previously were listed.

Summarizing this mass of literature in his thesis, Clark (1979) gave the following description of the effect of fire on bitterbrush. Possibly because of bitterbrush's extensive range, climatic and environmental conditions under which it grows, its response to burning is highly variable. Although most bitterbrush reproduces from seed (West 1968) it is commonly felt that fire destroys bitterbrush by removing both the existing stand and seed source. Most reports discussing the response of bitterbrush to burning are results of wildfires and are not well documented with respect to soil textural characteristics, soil moisture, phenological stage of development, weather conditions, or other environmental conditions which may influence post-fire recovery.

In California, Nord (1965) reported highly variable post-burn recovery, where 5 of 13 wildfires resulted in at least 5% sprouting while one burn exceeded 25% sprouting. Hormay (1943) noted that the only instance in which substantial sprouting (over 25%) occurred was a January wildfire in northeastern California. Hormay stated that hundreds of thousands of hectares of bitterbrush have been destroyed by fire. Countryman and Cornelius (1957) reported a complete absence of bitterbrush regeneration six years after a wildfire although it constituted 91% of the vegetative cover adjacent to the burn. Leopold (1950) emphasized the highly variable response by noting that in the Truckee River Canyon of eastern California logging and recurrent fires stimulated extensive growth of bitterbrush, but a few kilometers toward Reno fires seemed to eliminate bitterbrush.

Southern Idaho frequently is described as an area where bitterbrush sprouts well following fire. Blaisdell (1950, 1953) reported that burning destroyed sagebrush but bitterbrush sprouted. The year following burning, 49, 43, and 19% of burned plants sprouted on light, moderate, and heavy burns, respectively. These results occurred on basaltic soils in a 41 cm precipitation zone. Similar results were obtained at the U.S. Sheep Station near Dubois, Idaho where precipitation is 28 cm. At Dubois, nearly all the plants sprouted after a small burn of light intensity in the fall of 1945, but only 25% sprouted on a large burn of heavy intensity in the fall of 1947. Blaisdell and Mueggler (1956) burned or severed bitterbrush plants 5 cm above ground level in Idaho. They found that 50 and 72%, respectively, of burned and severed plants sprouted, and that sprouting occurred as late as 13 months after treatment. Also, mortality was high on sprouted plants. Of the burned plants that sprouted, 33% died within 12 months compared to 21% death among the severed plants. Clark et al. (1982) burned and clipped bitterbrush during fall and spring, under different soil moisture conditions on two sites in eastern Oregon. Sprouting after treatment was similar on the two sites, and burning resulted in greater mortality than clipping. Artificially watering plants before or after burning did not result in substantial reduction in mortality. Over-winter mortality of sprouts was high.

Driscoll (1963) found in Oregon that sprouting ability was related more to soil factors than to burn intensity. On northerly slopes, stands of bitterbrush found on loose, coarse-textured, nonstony soils without cinders or pumice had the highest frequency of sprouting. In one such area, 80% of the burned plants sprouted. Driscoll also noted that bitterbrush on these sites frequently layered, and plants which sprouted did so from the layer but never from the parent plant. Weaver (1957) observed in the ponderosa pine zone of Oregon, bitterbrush rapidly and heavily invaded after logging but was readily killed by fire. On areas where it dominated ground cover, it had not reestablished 18 months after wildfire.

Less information on burning response is available from other areas. In central and northern Utah, Blaisdell and Mueggler (1956) observed only limited sprouting. Daubenmire (1970) found bitterbrush in Washington nearly always is killed by steppe fires. In the Great Basin, Billings (1952) generalized that bitterbrush is eradicated by fire since it rarely rootsprouts in that region and its seeds are not particularly mobile.

The 1982 Symposium on Research and Management of Bitterbrush and Cliffrose in Western North America, held in Salt Lake City, presented many interesting viewpoints and some new data. Rice (1982) presented a fresh review of often cited literature on bitterbrush and concluded fire should be used in bitterbrush communities. Although some damage will occur, the uneven-aged stands that result are desirable. Contrary to other work in Washington, Driver (1982) observed 40 to 100% sprouting from spring burns in the central portion of the state. In central Oregon along the eastern edge of the southern Cascades, Martin (1982) reported bitterbrush mortalities of 77 to 100% for prescribed burns. Nineteen of 22 burns had mortalities higher than 97%. However, bitterbrush seedlings established quickly producing uneven-aged stands.

Conclusions on Bitterbrush

In reviewing the wealth of information on bitterbrush, no clear-cut answers emerge. The only consistent trend evident is the high level of variation in response. The following are trends that generally hold:

1. Spring burns are generally less damaging contrasted to fall burns.
2. Summer burns are very damaging.
3. Good soil moisture at the time of spring burning and for the first growing season is beneficial.
4. There is a tendency for plants to respond favorably at higher elevations.
5. Decumbant growth forms respond better to fire compared with columnar forms.
6. There is no evidence that fire intensity influences bitterbrush sprouting. However, fuel consumption

appears to have some correlation. Where consumption of fuel is high, mortality of bitterbrush is generally high, and seedling establishment is increased. Where consumption is low, bitterbrush mortality is reduced, and seedling establishment is reduced.

A final recommendation is to burn a small plot adjacent to the major burn being planned. This pilot burn should be at least one ha in size and contain a minimum of 50 bitterbrush plants that are marked with steel stakes. The burn should be conducted under the same prescription as the planned major burn. Marked plants should be observed closely for at least one growing season and through one winter. Measure everything accurately and in sufficient numbers to yield useful data. Although this technique will delay the major burn for at least 12 months, it will give a reasonable estimate of bitterbrush response for the area of interest.

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Threshold Requirements for Fire Spread in Grassland Fuels

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Abstract

During the spring months of 1981 through 1983, 118 test fires were attempted in three grassland species. Fires were set as line headfires or backfires under a wide range of fuel and weather conditions. Fires that carried across the fine fuel portion of the fuel bed array, irrespective of fireline intensity, were considered successful. Threshold weather and fuel requirements for successful fires, predicted by stepwise discriminant analysis, varied among species and were different for headfires than for backfires. For headfires, success was best predicted by air temperature, and green and down litter fuel loads for buffalograss (Buchloe dactyloides (Nutt.) Engelm.), and by down litter moisture content, solar intensity, silica content, and low heat of combustion for tobosagrass (Hilaria mutica (Buckley) Benth.). Headfire success of weeping lovegrass (Eragrostis curvula (Schrad.) Nees) was not predictable with any of the measured variables due to the low number of failed fires. For backfires, success was best predicted by down litter fuel load, air temperature, windspeed, and relative humidity for tobosagrass, and by the ratio of green:standing litter fuel, relative humidity, and time since a 20 mm precipitation event for weeping lovegrass. Buffalograss backfire success could not be predicted due to the lack of successful fires. When data from all species were combined, headfire success was best predicted by down litter fuel moisture content, air temperature, ratio of green:total fuel, standing litter fuel moisture content, and standing litter ash content. Combined data for backfires predicted success from total fuel load, total fuel moisture content, windspeed at 0.5 m, and relative humidity. Although precision declined, commonly measured variables also predicted successful fires: total fuel load, total fuel moisture content, and windspeed for headfires, and total fuel load, total fuel moisture content, and relative humidity for backfires.

Introduction

Preburn planning and fireline preparation, the most expensive phases of prescribed burning on public rangelands, are usually completed before ignition is attempted. If the burn then fails due to insufficient fuel, inadequate fuel distribution, improper fuel or weather conditions, or some combination of these problems, the economic advantage of prescribed burning over alternate treatments is lost. Further, the failure may result in a substantial delay of implementing an alternate treatment. It is therefore useful to know early in the planning process if a site should be burned, can be burned, how to burn it, and the probability of success. The fire environment of a site should be evaluated early in the planning process, and must be considered in an adequate burn plan.

Fire environment is the surrounding conditions, influences, and modifying forces of topography, fuel, and air mass that determine fire behavior (Wein and MacLean 1983). Many variables contribute to the fire environment, and effects of the variables acting simultaneously may be synergetic, synergistic, or antagonistic. Ultimately, the fire either succeeds or fails. Van Wagner (1983) described three universal principles that apply to successful fires and two of the principles are paramount in prescribed burns. First, there must be sufficient fuel of appropriate size and arrangement. Second, the fuel must be of sufficient dryness to support a spreading combustion reaction. If these two conditions are satisfied, fire will spread with site factors and weather conditions controlling fire behavior. Unfortunately, little research in grassland fuels has been directed toward Van Wagner's principles. Grassland fire behavior and fireline intensity-flame length relationships are presented elsewhere (Clark 1983); the purpose of this paper is to describe results of a study designed to isolate fuel and weather thresholds for successful fires in grasslands.

Methods

Head and back test fires were attempted during the spring months of 1981 through 1983, in uniform stands of three grass species, under

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Table 1. Range of selected variables used in discriminant analyses of headfires and backfires in three grass fuels.

Fire Environment Variable	Buffalograss	Tobosagrass	Weeping lovegrass
Down litter fuel load (kg ha^{-1})	160 to 517	772 to 4469	400 to 3792
Standing litter fuel load (kg ha^{-1})	478 to 1739	1892 to 5136	1572 to 6674
Green fuel load (kg ha^{-1})	188 to 470	50 to 1387	0 to 1133
Total fuel load (kg ha^{-1})	975 to 2401	2791 to 8939	3308 to 8340
Down litter fuel moisture content (%)	1.0 to 104.2	5.1 to 23.8	0.8 to 174.3
Standing litter fuel moisture content (%)	1.0 to 25.6	4.5 to 37.8	2.3 to 181.4
Total fuel moisture content (%)	24.1 to 57.6	14.6 to 59.3	7.3 to 180.8
Ratio green fuel load:standing litter load (fraction)	0.15 to 0.92	0.01 to 0.35	0 to 0.33
Ratio green fuel load:total litter load (fraction)	0.13 to 0.47	0.01 to 0.08	0 to 0.18
Standing litter ash content (%)	13.1 to 19.3	8.6 to 13.7	4.8 to 11.4
Green fuel silica content (%)	3.7 to 14.3	4.6 to 13.0	1.8 to 9.5
Standing litter low heat content (kJ kg^{-1})	14766 to 18457	16309 to 17831	16441 to 18434
Time since 20 mm precipitation event (days)	4 to 173	5 to 172	2 to 159
Air temperature at ignition (°C)	8.0 to 31.1	4.7 to 32.8	-2.8 to 30.3
Relative humidity at ignition (%)	14 to 71	11 to 97	12 to 95
Average windspeed at 0.5 m (km hr^{-1})	3.6 to 26.5	3.1 to 25.9	2.7 to 20.4
Average windspeed at 2.0 m (km hr^{-1})	4.7 to 32.1	3.7 to 53.1	6.1 to 39.3
Solar intensity at ignition (W m^{-2})	10 to 950	44 to 940	100 to 950

a wide variety of fuel and weather conditions (Table 1). Attempts included 19 fires in buffalograss (*Buchloe dactyloides* (Nutt.) Engelm.), 55 in tobosagrass (*Hilaria mutica* (Buckley) Benth.), and 44 in weeping lovegrass (*Eragrostis curvula* (Schrad.) Nees). Buffalograss is a short, finely-divided, stoloniferous grass; tobosagrass is a mid-height, coarse, rhizomatous grass; and weeping lovegrass is a tall, fine-leaved, introduced bunchgrass. Fires that carried across the fine fuel portion of the fuel bed, irrespective of fireline intensity or fuel consumed, were considered successful.

All fires were conducted in uniform fuels on level terrain to minimize effects of fuel discontinuity and slope. In each test fire, central areas that were 20 m (factors for conversion to English equivalents are presented in an Appendix) or longer in buffalograss, 30 m or longer in the other species, and at least 12 m wide were used for fire behavior measurements; fuel data (Table 2) were collected around the perimeter of each central area to minimize fuel bed disturbance.

Fuel data were determined by clipping to within 1 cm of the soil surface a sufficient number of quadrats on each test fire to estimate fuel load within 20% of the mean at 90% confidence. This was usually 10, 0.25-m^2 rectangular quadrats in weeping lovegrass and 15, 0.06-m^2 rectangular quadrats in buffalograss and tobosagrass. Each quadrat load was separated into green, standing litter, and down litter components and oven-dried to constant weight at 60°C. Sub-samples were weighed to the nearest g in the field immediately after collection, oven-dried at 60°C to constant weight, then reweighed to obtain fuel moisture content on a dry weight basis. Recorded dry weights also provided ratios of green fuel: standing litter, and green fuel:total litter.

Fuel chemistry was determined using standard laboratory procedures. Analyses completed were low heat content determined in a Parr Adiabatic Bomb Calorimeter (Parr Inst. Co., n.d., Van Wagner 1972), and ash, silica, and phosphorous (AOAC 1980). For the 1981 test fires, all three fuel load components were analyzed for each fire. During 1982 and 1983, three to five samples were randomly selected for each component of each grass species and analyzed to verify the 1981 data.

Surface area:volume ratios were determined for each species (Clark and Wright 1981). Packing ratios, calculated as the ratio of bulk density:particle density (Rothermel 1972), varied little within species, thus, single values of 0.004130, 0.003461, and 0.005552 were used for buffalograss, tobosagrass, and weeping lovegrass, respectively.

Several weather variables were also measured (Table 2). Air temperature and relative humidity were determined within three min of ignition with a standard belt weather kit (Anon. 1959) sling psychrometer. Solar intensity was determined with a LI-COR LI-185 Radiometer and LI-200S Pyranometer Sensor (LI-COR, n.d.). Windspeed at 0.5 and 2.0 m was determined with totalizing anemometers (Clark et al. 1981). The on-site weather measurements were made 20 m upwind of the fire. Also, total precipitation since 1 October and 1 January, and the time since a 20 mm precipitation even occurred were obtained from National Weather Service records (NOAA 1981-1983) supplemented by local gages and records.

Data was analyzed using the BMDP7M computer program (Jennrich and Sampson 1981) to perform stepwise discriminant analyses that categorized test fires into success or failure groups. Dual discriminant functions were combined (Morrison 1976) into single equations to simplify use. Jackknifed classifications were used because success or failure of each fire was grouped according to classification functions computed from all data except the fire being classified. This classification method is more restrictive than normal classification and provides a more realistic estimate of the ability of predictors to discriminate among groups (Tabachnick and Fidell 1983).

Results and Discussion

All Variables Included

Lachenbruch (1975) suggested that when a large number of independent variables are used in stepwise discriminant analysis, only three to five variables can be safely selected without introducing noise. Therefore an attempt was made to select significant functions based on these criteria.

The use of all variables (Table 2) generated a function that correctly grouped 86% of buffalograss headfires. Success was partially dependent on fuel load; however, the coefficient for green fuel load was negative and twice as large as that for the down litter fuel load. This suggests that fires occurring after plant growth initiation in the spring, or after regrowth in the fall, are progressively less likely to succeed.

Table 2. List of variables considered in discriminant analyses of head and back test fires in three grass fuels.

X_1	=	Time since 20mm precipitation even occurred (days)
X_2	=	Total precipitation since 1 January of fire year (mm)
X_3	=	Total precipitation since 1 October of preceding year (mm)
X_4	=	Fuel load, standing litter (kg ha^{-1})
X_5	=	Fuel load, green (kg ha^{1})
X_6	=	Fuel load, down litter (kg ha^{1})
X_7	=	Fuel moisture content, standing litter (%)
X_8	=	Fuel moisture content, green (%)
X_9	=	Fuel moisture content, down litter (%)
X_{10}	=	Fuel bed depth (cm)
X_{11}	=	Ratio green fuel load:standing litter load (fraction)
X_{12}	=	Ratio green fuel load:total litter load (fraction)
X_{13}	=	Windspeed at 0.5m above soil surface, average (km hr^{-1})
X_{14}	=	Windspeed at 2.0m above soil surface, average (km hr^{-1})
X_{15}	=	Solar intensity at ignition (W m^{-2})
X_{16}	=	Ash content, standing litter (%)
X_{17}	=	Ash content, green fuel (%)
X_{18}	=	Ash content, down litter (%)
X_{19}	=	Phosphorus content, standing litter (%)
X_{20}	=	Phosphorus content, green fuel (%)
X_{21}	=	Phosphorous content, down litter (%)
X_{22}	=	Silica content, standing litter (%)
X_{23}	=	Silica content, green fuel (%)
X_{24}	=	Silica content, down litter (%)
X_{25}	=	Low heat content, standing litter (kJ kg^{-1})
X_{26}	=	Low heat content, green fuel (kJ kg^{-1})
X_{27}	=	Low heat content, down litter (kJ kg^{-1})
X_{28}	=	Air temperature at ignition (°C)
X_{29}	=	Relative humidity at ignition (%)
X_{30}	=	Fuel particle surface area:volume ratio (cm^2 cm^{-3})
X_{31}	=	Packing ratio, fuel bulk density: particle density (cm^3 cm^{-3})
X_{32}	=	Fuel load, total (kg ha^{-1})
X_{33}	=	Fuel moisture content, total (%)

Tobosagrass headfire success was influenced by chemical factors more than was buffalograss, required more variables, and was more difficult to predict. The inclusion of solar intensity, green fuel silica content, and standing litter low heat of combustion resulted in a discriminant equation that, while useful in understanding fire behavior, was not practical for field use. Therefore, the reader who is interested in equations based on all variables for these species is referred to Clark (1983). When data from all headfires on all species was pooled, the ratio of green fuel load:total litter fuel load became more important. Implicit in this result is the contention that, irrespective of the amount of standing or down litter available for combustion, the proportion of green fuel in the fuel bed matrix can influence the success of fires in these species.

In contrast to buffalograss and tobosagrass headfires, weeping lovegrass headfires were usually successful. The only failure in 25 attempts occurred the morning following a 5 cm snowfall. Successful fires were conducted with air temperature -3°C and relative humidity 93%. When total fuel moisture exceeded 30%, combustion was not complete; on one fire a 15 cm stubble remained after burning although the fire carried through standing litter. The weeping lovegrass stands in which most of the fires were conducted were decadent, ungrazed pastures with abnormal accumulations of dead material. Therefore, the discriminant equation in which data from all species was pooled may not be applicable to prescribed burning in properly managed pastures.

Backfires are rarely used in large-scale prescribed burning except to create blacklines, minimize threat of escape near downwind fire perimeters, and to reduce flame length under canopies. Thus, little information is available in the literature concerning requirements for backfire success. In this study, variables commonly measured on prescribed fires including air temperature, relative humidity, windspeed, fuel load, and fuel moisture content consistantly predicted success of backfires, even when all variables were used. These predictions are discussed below.

Five Variable Restrictions

Fire managers do not routinely have access to equipment, training, or time, to measure all the variables in Table 2. Therefore, five easily measured variables, including total fuel load, total fuel moisture content, air temperature, relative humidity, and windspeed at 2.0 m were used to determine whether fire success or failure could be correctly classified and thus provide a field-oriented method to predict fire success. Separate analyses were made for headfires on each species ad for all species combined (Table 3).

With headfires, buffalograss success or failure was not predictable (P=.05), and weeping lovegrass data did not provide a reasonable result because only one headfire failed. Success or failure of tobosagrass was predictable, however. Only two variables, total fuel load and total fuel moisture content, provided an 81% classification. The influence of tobosagrass data also allowed the combined data for all three species to correctly classify 88% of the successful and

Table 3. Discriminant equations, based on five commonly measured variables, for head and backfires in three grass species. Calculated values greater than 0 predict success at P = .05.

Headfires	Prediction Equation[1]	Correct Prediction (%)
Buffalograss[2]		
Tobosagrass	$+1.0 + 0.001237X_{TFL} - 0.044305X_{TFM}$	81
Weeping lovegrass[3]		
All species	$+1.0 + 0.000437_{TFL} - 0.058059X_{TFM} - 0.026498X_{WS}$	85
Backfires		
Buffalograss[4]		
Tobosagrass	$+1.0 + 0.000377X_{TFL} - 0.037166X_{WS} - 0.049282X_{RH}$	71
Weeping lovegrass	$-1.0 + 0.000130X_{TFL} + 0.016795X_{AT} - 0.003392X_{TFM}$	79
All species	$-1.0 + 0.000431X_{TFL} - 0.013111X_{TFM} - 0.028647X_{RH}$	80

1 X_{TFL} = total fuel load (kg ha^{-1}), X_{TFM} = total fuel moisture content (%), X_{RH} = relative humidity (%), X_{WS} = windspeed at 2.0 m (km hr^{-1}), and X_{AT} = air temperature (°C).

2 No statistically significant (P = .05) equation generated.

3 No reasonable equation generated due to low number of failures

4 No buffalograss backfires were successful

failed fires. The range of fuel loads and fuel moisture contents used to generate equations for combined species was 875 to 8939 kg ha^{-1} total fuel load, and 7 to 181% total fuel moisture content.

Backfire success or failure was predictable for tobosagrass and weeping lovegrass, but not buffalograss (Table 3). Combined data indicated that backfire success was predictable using total fuel load, total fuel moisture content, and relative humidity.

Use of Equations

The discriminant equations are applicable to the species listed, and may be applicable to other species with similar morphological characteristics. Due to the diverse morphological differences among the three species, equations for "all species" may be applicable to mixed grassland communities with uniform, contiguous growth patterns on level terrain. Equations determined from five easily measured fuel and weather variables are easily applied to field use and should provide guidelines for planning prescribed burns. However, extrapolation beyond the range of variables used in this study should be done with caution. Some calculated examples are listed in Table 4. Note, for example, that less combustible combinations (e.g., total fuel moisture content of 20% with fine fuel load of less than 250 kg ha^{-1}) will lessen the probability of successful fire spread.

It should be emphasized that erratic fire behavior leading to fire whirls or spotting, or horizontal fuel discontinuity, were not addressed in this study. Also, fires were ignited as line fires with drip torches; helitorch or terratorch ignition would probably create spreading fires that would be unsuccessful with drip torches.

Table 4. Examples of fuel and weather conditions that will result in fires that carry across fine (0.3 cm) fuel about 80% of the time. These values assume relatively uniform fuel distribution on level terrain.

Fuel Moisture Content (%)	Windspeed at 2 m (km hr-1)	Fine Fuel Load (kg ha-1)
20	10	2000
10	10	1000
10	10	500
10	5	250
10	10	200

Sagebrush Overstory

The contribution of a shrub overstory to fire success was not evaluated in this study. Other research (Britton et al. 1981) indicates that a nonlinear relationship exists between required fine fuel load and shrub canopy cover when a shrub component is present. Until the thresholds presented here are tested with a shrub overstory, the presence of shrubs should reinforce predictions of success provided other required conditions are met.

Management Implications

Test fires in this study were "successful" when they carried across the fine fuel portion of the fuel bed. Prescribed burns in shrub-grass mixtures may have more stringent objectives such as consuming sagebrush skeletons, scorching upper crowns of juniper (*Juniperus* sp. L.) trees, or killing weakly sprouting shrubs such as 3-tip sagebrush (*A. tripartita* Rydb.). Obviously, fires that are more intense or more severe are required to meet these objectives; generally, fireline intensity and reaction intensity can be increased by using heavier fine fuel loads, warmer and drier weather conditions, and possibly faster windspeed. Conversely, severity can be increased with slower fires or by burning during periods of physiological stress in plants, burning during an unfavorable season, burning during drought, or burning frequently. Fire intensity and fire severity should not be confused.

Also, reliable estimates of fine fuel loads, moisture contents, and other fuel and weather variables are necessary for the relationships described here to work. Windshield surveys are simply not adequate, and statistically reliable measurements, that include fuel load by size class,fuel moisture content by size class, and canopy cover by size class are essential. Fuel loading and cover should be measured even before the burn plan is written to allow the fire behavior specialist to determine if the site can be burned, and the best approach to burning it. These data can also help range managers, botanists, wildlife biologists, foresters, or other specialists determine whether the site should be burned. Inventory needs and techniques are discussed by Neuenschwander (these proceedings); however, the need for sampling of fuel, and weather variables that are necessary to evaluate fire behavior and burn success cannot be overstated.

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Appendix

Factors (approximate) to convert selected SI units to English equivalents.

SI Unit	Multiply By	To Obtain English Units
Length/ Distance		
mm	0.0394	inches
km	0.6214	miles
m	0.0497	chains
Area		
m^2	10.7639	$feet^2$
ha	2.4710	acres
Mass		
g	0.0353	ounces
kg	2.2046	pounds
Temperature		
°C	1.8°C+32	°F
Energy		
J	0.0010	Btu
cal	0.0040	Btu
Mixed Units		
cal g^{-1}	1.8000	Btu lb^{-1}
cal $m^{-1}s^{-1}$	0.012	Btu ft^{-1} sec^{-1}
g m^{-2}	8.9220	lb $acre^{-1}$
m min^{-1}	2.8922	chains hr^{-1}
kg ha^{-1}	0.8922	lb $acre^{-1}$
kg m^{-3}	0.0624	lb $foot^{-3}$
kJ kg^{-1}	0.4302	Btu lb^{-1}
km hr^{-1}	0.6214	miles hr^{-1}
kW m^{-1}	0.2891	Btu ft^{-1} sec^{-1}
m s^{-1}	2.2369	miles hr^{-1}

Note: Data given in English units can be converted to SI units by multiplying by the conversion factor reciprocal.

Fire Effects and Revegetation on Juniper-Pinyon Woodlands

Richard Everett and Warren Clary

Abstract

Fire is a natural phenomenon in juniper-pinyon-sagebrush ecosystems, but fire may now produce plant communities different from what occurred before the influence of man. Lack of indigenous understory species and the invasion of alien annual weeds now create new postfire community types. Mechanically created firebreaks disturb the soil surface and set back plant succession. We need to know if chained and seeded strips can be used as firebreaks in closed stands to improve burn success and postfire plant response. Adverse impacts of burning slash on soil nutrients, understory productivity, and watershed characteristics make this practice of dubious value. The "Initial Floristics" successional model typifies postfire succession. Most species in the sere are present on the site shortly after disturbance. Many annual and perennial forb species have only a short period in which to recharge soil seed reserves and provide seed to other disturbed sites. Seeding burns may be the only way to restore the grass successional stage to many of our overgrazed woodlands. Broadcast seeding success following fire has been variable, but could be improved by considering slope, aspect, and elevation effects. Postfire cultural treatments drilling and chaining increase seeded species establishment over that of aerial broadcast seeding alone. Seeded plant establishment was directly proportional to the intensity of postharvest cultural treatment. In a wildfire seeding study aerial seeding produced the least seeded plant cover (0.1 percent) and double drilling the greatest (13.8 percent).

Introduction

The juniper-pinyon and associated sagebrush ecosystems are extensive in the Intermountain West. Utah juniper (*Juniperus osteosperma*) a major component of the woodland, occurs across Nevada, southern Idaho, Utah, western Colorado, and western Arizona (Arno 1971). Utah juniper occurs in pure stands and in an array of species mixtures. Other species commonly occurring as codominants include pinon (*Pinus edulis*), singleleaf pinyon (*Pinus monophylla*), one seeded juniper (*Juniperus monosperma*), and numerous sagebrush (*Artemesia* sp.) species. Juniper invasion into adjacent sagebrush communities and the subsequent forage depletion will not be curtailed by grazing management alone; some form of tree control is required to reduce tree competition (Jameson and Reid 1965).

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Fire is a natural phenomenon in juniper - pinyon - sagebrush ecosystems and has been recommended to control juniper (Arno 1971). Past grazing abuse and fire suppression in juniperpinyon woodlands have prolonged tree dominance (Tausch et al. 1981). Soil seed reserves and remnant plants have declined and seriously lowered the capability of the understory to respond to fire (Everett and Sharrow 1983). Postfire succession may not lead to the desired plant community (Clary et al. 1974, Everett and Ward 1984). We need improved predictability of natural response and improved methods of seeding burns not capable of providing a desirable understory response. We have many juniper stands that could be burned, but they require seeding to improve forage and exclude alien annuals. Postfire dominance by highly flammable annual grasses presents a serious resource problem (Favre 1942, Mutch 1967).

Burning Utah Juniper

Woodlands contain valuable wood resources that should be burned only after alternative management strategies have been considered. Where possible tree products such as cord wood, Christmas trees, and fence post material should be removed prior to burning. To get a clean controlled burn, specific weather conditions must be met that may only occur a few days a year or not at all (Pase and Granfelt 1977). Burn prescriptions have been developed for juniper in open grasslands, shrublands, pinyon-juniper mixed stands, and closed juniper communities (Arnold et al. 1964, Truesell 1969, Ralphs et al. 1975, Bruner and Klebenow 1979, and Wright et al. 1979). Basically, as understory and stand density increase, stands become easier to burn. The ability to burn sagebrush and juniper-pinyon woodlands effectively is keyed to the fuel loads of understory species (Brown 1982, Bruner and Klebenow 1979, Wright et al. 1979). Burning ease, understory fine fuels, and potential for natural plant response would appear greatest in lightly grazed ecotonal areas of big sagebrush (*Artemesia tridentata* ssp. *tridentata*), pinyon, and juniper. Ecotonal areas often occur in drainage bottoms and invaded riparian zones. Burn success has been high and adjacent closed stands serve as natural firebreaks (Bruner and Klebenow 1979). Seeding may not be required for adequate plant response.

Firebreaks and Plant Response

Firebreaks are often required for prescribed fire and wildfire burns. Mechanically built firebreaks disturb soils and set plant succession back to its earliest stages. The areas are often seeded. Ralphs et al. (1975) recommended firebreak lines 30 m wide in juniper and suggested numerous small ignition plots within a burn. Davis (1976) recommended the use of chaining to establish firelines. We need to know if seeded chained areas would act as green fuelbreaks and as supplementary seed sources for the subsequent prescribed burn. Economics of chained firebreaks may be favorable because (1) chaining costs could be spread over the entire chainedburned area, (2) wide firebreaks created by chaining may reduce risk of fire escape and allow burning under more severe conditions, and (3) seeded chained areas could serve as a supplementary seed source for the subsequent burn.

Bruner and Klebenow (1979) encouraged the use of natural firebreaks because of economics and the lack of soil disturbance. Changes in vegetation, snow pockets, and topography have been used as firebreaks and to give a patterned appearance to burns (Beardall and Sylvester 1976, Blackburn and Bruner 1979).

Burning Juniper Debris Piles

Burning slash removed 90 to 95 percent of small trees left after chaining and windrowing (Pase and Granfelt 1977). Burning windrows intensified the heat load on the soil surface and raised soil temperatures to 55°C in the interspace and 288°C under the slash piles (Gifford 1981). Total soluble salts, P, K, N, and C increased in surface soils the first year following burning of juniper slash piles, but the effect diminished by the second year (Gifford 1981). Soluble salts rapidly leached from the surface to a 5 to 10 cm depth. Soils in burned debris sites have reduced infiltration (Arnold et al. 1964, Buckhouse and Gifford 1976a). Overland flow from burned areas contained greater quantities of K and P than unburned areas (Buckhouse and Gifford 1976b). A light cover of slash increased forage production of understory species (Arnold et al. 1964). Burning of slash appears unwarranted because of the loss of plant and soil nutrients, loss of ground cover, reduced understory response, and potential decline in watershed quality.

Understory Response to Fire

Fire in juniper woodlands occurs over an array of microsites that vary in soil and plant characteristics. Tree and shrub species alter soil chemistry (Barth 1980, Osagrande 1978), and create understory patterns (Johnsen 1962). Plant patterns create heterogenous fuel loadings impacted by firebreaks can be extensive but are across the soil surface and provide an array of burn conditions. Klebenow et al. (1977) found maximum surface soil temperatures to be 200°C higher under shrubs than in the interspace in a prescibed pinyon-juniper burn. Burning trees release large amounts of energy (120 to 1064 x 10^3 calories) to the soil surface (Gifford 1981). The soil under burned tree crowns is often devoid of litter, understory plants, and soil seed reserves. Initial response is often less in burned duff than adjacent interspace.

Vegetation changes with soil type, thus we can expect postfire response to vary across soil boundries (Mason et al. 1967). Thatcher and Hart (1974) found postfire plant response varied among sites on the same burn because of the presence or absence of a vesicular soil surface. Sites with a vesicular crust did not produce a grass stage during the sere and those without the crust did.

Postfire Succession

Postfire succession has been presented as an orderly series of plant form changes from annuals to eventual tree dominance (Arnold et al. 1964, Erdman 1970, Barney and Frischknect 1974), as depicted by the "Relay Floristics" model (Egler 1954). Postfire response following prescribed burns and wildfires in juniper-singleleaf pinyon stands of Nevada suggests the "Initial Floristics" model is more appropriate (Everett and Ward 1984). In this model, most sere species are present immediately after disturbance occurs. Given a burn of sufficient size, several successional stages from annual to shrub dominance may be present (Everett and Ward 1984, Fig.1). Initial postfire vegetation has been dominated by shrubs on several occasions (Thatcher and Hart 1974, Clary et al. 1974, Everett and Ward 1984).

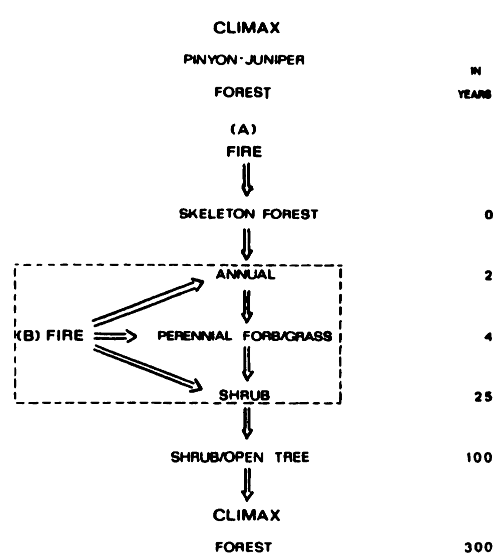

Figure 1.--Successional model of pinyon-juniper woodland following fire (Erdman 1970). (B) Multiple entrance points into the successional model (Everett and Ward 1984).

Predictability of postfire response was difficult because of unknown soil seed reserves, immigrating propagules, and postfire precipitation. Multiple successional pathways can create problems if specific plant communities are desired. Fortunately, consistencies in the occurrence of plant forms with aspect and elevation, and the rapid return of prefire species improved predictability somewhat (Koniak 1985). Two thirds of the preburn species returned within 5 years following prescribed burns in mixed pinyon-juniper stands (Everett and Ward 1984). Koniak (1985) found in the Great Basin that shrubs and perennial grasses were the dominant understory forms in burns on mesic high elevation north and east aspects, and annual grasses and forbs predominated on low elevation south and west aspects.

On some sites annual (i.e. Nicotiana attenuata), biannual (i.e. Argemone munita), and some perennial species (i.e. Sphaeralcea munroana) have only a short time in which to recharge soil seed reserves and provide seed to other disturbed sites (Fig. 2). Recharge of soil seed reserves is important because germinating seed, in combination with remnant plant growth, are often the source of initial plant response to fire. These short term cycles and the longer cycles of shrubs and grasses (Barney and Frischknect 1974)) are important considerations in determining wildlife and grazing benefits from prescribed burns.

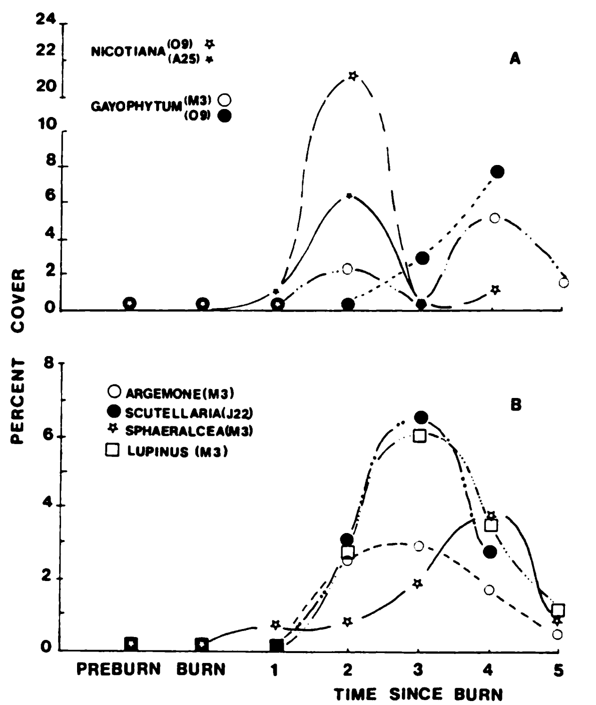

Figure 2.--Cover of annuals (A) Nicotiana attenuata and Gayophytum ramosissimum, and perennials (B) Argemone munita, Scutellaria nana, and Sphaeralcea munroana, and Lupinus caudatus by growing season following burning. Data for burns on April 24 (A25), May 3 (M3), June 22 (J22), and October 9 (O9) are presented (Everett and Ward 1984).

Fire suppression and grazing have prolonged tree dominance, reduced understory vigor, and altered understory composition (Tausch et al. 1981). Jameson et al. (1962) reported increased grass cover in ungrazed and often burned Utah juniper stands on Fishtail Mesa vs. adjacent unburned "mainland" grazed areas. Schmutz et al. (1976) found 88 plant species in ungrazed stands and only 38 species on grazed pinyon-juniper stands in Arizona. Grasses made up 36 percent of the vegetation on ungrazed sites and only 6 percent on grazed sites. Livestock grazing has apparently removed many species from the understory. Fire response on grazed and ungrazed sites would be floristically different.

Fire responsive alien annual grasses such as cheatgrass (Bromus tectorum) and alien annual forbs such as Russian thistle (Salsola pestifer, Salsola kali), tansymustard (Descurania sophia), and tumblemustard (Sisymbrium altissimum), are important floristic components of early successional stages on burned juniper woodlands (Erdman 1970, Barney and Frischknect 1974, Stager 1977). Postfire vegetation on 10 of the 11 burns reported in these studies contained cheatgrass. Percent cover of cheatgrass declined from 13 percent after 3 years to a trace after 17 years, but frequency remained high (22 percent) for several decades. Fire may be a natural phenomenon in the juniper woodland, but current postfire response may be drastically different from what occurred prior to the impact of man.

Seeding Wildfires and Prescribed Burns

Broadcast seeding has been used to get seed on the ground at minimal initial cost but seeding success has been variable. Elevation, aspect, and postseeding precipitation impact broadcast seeding success. Koniak (1981) found seeding success was greater on higher elevations and on north and east aspects in pinyon-juniper stands of the Great Basin. We need to adapt our seed mixtures to topographic differences and the problem of no seed coverage. Until adapted seed mixes can be derived, our best chance to increase seeding success is through postfire cultural treatments. The tentative results from an evaluation of postfire cultural treatments are provided below.

The Canyon Mountain Postfire Seeding Trials

Clary (unpublished data) evaluated the impact of aerial seeding, chaining, drilling, and land imprinting on the establishment of seeded perennial grasses and forbs. Trials were conducted in burned juniper-pinyon-sagebrush mixed communities. The study site was the 1981 Little Oak Creek and Clay Springs wildfires in the Canyon Mountains of western Utah. The predominately wheatgrass (Agropyron sp.) seed mix was applied at approximately 6 to 10 kg/ha. Because the burn site was heterogenous, all

Table 1.--Evaluation of Postfire Cultural Treatments on Establishing Cover of Seeded Perennial Grasses and Forbs (Clary, unpublished data).

Burn	Treatment 1	Treatment 2	Percent cover[1] T 1	Percent cover[1] T 2	Difference Percent
O[2]	Double Rangeland Drill	Single Rangeland Drill	13.8	7.8	45
O	Land Imprinter	Single Rangeland Drill	9.8	5.5	44
O	Single Rangeland Drill	Single Chained	3.6	.7	80
O	Double Chained	Single Chained	1.6	.8	50
C	Double Chained	Single Chained	7.6	4.7	39
C	Single Chained	Aerial Seeded Only	4.7	.1	98

1/ T 1: Treatment 1, T 2: Treatment 2.
2/ O: Little Oak Creek Burn, C: Clay Creek Burn.

treatments could not be compared simultaneously. An approximate ranking of treatment success was made through a series of paired comparisons (Table 1). Seeded grass and forb cover was directly related to the intensity of treatments applied. The double Rangeland Drill treatment (seeded twice) had the highest seeded plant cover (13.8 percent) and aerial seeding the lowest (0.1 percent). The more times an area was subjected to the same cultural treatment (drilling or chaining) the greater was the seeding success. Cover of annual grasses declined as cover of desirable seeded species increased.

SUMMARY

Fire effectively reduces Utah juniper competition and releases understory species. Overgrazing, prolonged tree dominance, and the presence of alien annuals now produces initial postfire communities different from that which occurred prior to European mans influence. Opportunities exist for increasing forage production through prescribed burning, but many stands will require seeding. Burning without successful seeding has the potential of creating alien annual grass communities that are extremely flammable and resistant to normal succession. Seeding may be the best way to get perennial grasses back into woodland succession and to reduce potential fire danger in this altered ecosystem. Broadcast seeding is chancy at best, but we may improve seeding success if we adapt seed mixtures to differences in aspect, elevation, and soil, and select species that require minimal seed coverage. Postfire cultural treatments chaining and drilling increased establishment of seeded species over that of aerial seeding alone. Undesirable annual grass cover is inversely related to seeding success.

The big sagebrush-juniper-pinyon ecotone and invaded riparian habitat appear to be opportune sites to conduct prescribed burns. Understory is present in adequate amounts to carry the fire and may provide an adequate postfire response. Adjacent closed Utah juniper stands provide a natural firebreak. In closed stands, we need to evaluate combining seeded chaining treatments for firebreaks, with burning of enclosed areas. Perhaps we can increase burn success and postfire plant response. Where practical, fuel wood, Christmas trees, and fence post materials should be removed prior to burning.

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Fire Effects and Application of Prescribed Fire in Aspen

James K. Brown

Abstract

The influence of low- and high-intensity fire on successional patterns of climax and seral aspen (Populus tremuloides) is discussed. Mechanisms of aspen regeneration and how fire affects these mechanisms are reviewed. Sucker densities following fire and the influence of fire severity and intensity on sucker response are discussed. Understory response to fire and grazing is summarized from the literature. Appropriate land management and fire objectives are suggested for maintaining the aspen forest. A vegetation-fuel classification is described that recognizes three classes where aspen dominates: aspen-shrub, aspen-tall forb, and aspen-low forb. Two classes are recognized in mixed conifer-aspen forests: mixed-shrub and mixed-forb. Fuel loadings, fire intensities, and probabilities of successful prescribed fire are described. The aspen-shrub class is the most flammable, and aspen-low forb and mixed-forb classes are the least flammable. Flammability is altered by slope, grazing intensity, curing of herbaceous vegetation, quantities of downed woody material, crown closure, and pocket gopher activity. Methods of predicting and estimating live fuel moisture contents are described.

Fire Effects and Application of Prescribed Fire in Aspen

Aspen (Populus tremuloides) is widely distributed throughout North America. It occupies about 2.86 million ha in the western United States (Green and Van Hooser 1983). Aspen forests provide wood products but are especially valued as wildlife habitat, for grazing by domestic livestock, as sources of water, and for esthetics and recreation (DeByle 1978). Fire has played an integral part in the development of aspen forests. This paper discusses the effects of fire in aspen forests and provides information on how to plan the use of prescribed fire to maintain these forests.

Successional Patterns

Aspen exists as both a climax and seral species but is seral on the majority of sites, eventually to be replaced by conifers (Mueggler 1976). On climax aspen sites, frequent fires can maintain a grass-forb community with aspen suckers confined to the shrub layer (Crane, unpublished report). Infrequent fires produce varying effects on stand structure. Low-intensity fires cause thinning and encourage an all-aged condition. High-intensity fires result in new even-aged stands.

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Replacement of seral aspen by conifers is gradual and may take 200 to 400 or more years (Bartos et al. 1983), depending on the potential for establishment and growth of conifers. This successional process reduces forage production from about 780 kg/ha to 225 kg/ha (Kranz and Linder 1973, Reynolds 1969), reduces water yields from about 50 area cm to 40 cm (Jaynes 1978, Gifford et al. 1983, 1984), and diminishes vegetation diversity and habitat for many species of wildlife (DeByle, in press).

Fire frequency and intensity in mixed aspen and conifer stands influence species composition and stand structure and can affect rate of succession. High-intensity fires generally replace stands, thus setting succession back to the herbaceous stage (Fig. 1). During the following immature tree stage, aspen dominates the overstory and conifers may become established in the understory. In the mature stage, conifers may join aspen in the overstory or occur only in the understory. If succession continues without fire, aspen will eventually be crowded out. Then fires will be followed only by new stands of conifers.

Response to low-intensity fires depends partly on the conifer species present. Generally, low-intensity fires will thin aspen stands; then aspen suckers and conifer seedlings will fill in the openings (Fig. 1). Intolerant species such as ponderosa pine and Douglas-fir often will survive low-intensity fires that will kill aspen. Thus they may be favored over aspen in the mixed stands. Tolerant species such as spruce and fir are unlikely to support low-intensity fire. When low-intensity fire occurs, however, they are readily killed, thus creating openings for new trees.

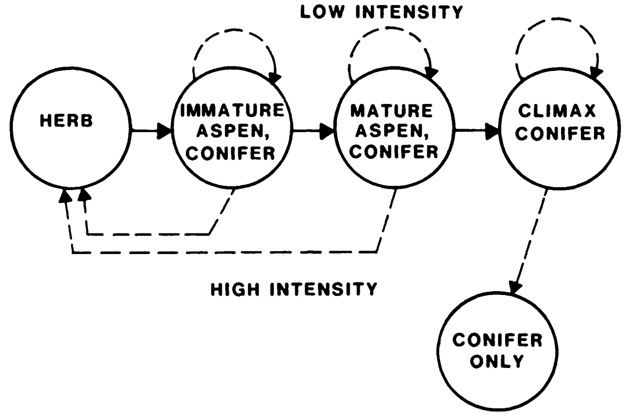

Fig. 1. Successional pathways involving seral aspen and conifers affected by low- and high-intensity fire.

Methods that have been used to interrupt succession include (1) prescribed fire, (2) herbicides, and (3) clearcutting. Fire frequently is the only economically and environmentally acceptable tool for retaining aspen.

Prescribed fire can be used to accomplish several objectives (Brown and DeByle 1982):

1. Set back plant succession.

2. Stimulate aspen regeneration from root suckers.

3. Temporarily increase production of herbaceous vegetation and shrubs.

4. Provide diversity in both age and size classes of aspen in the forest.

5. Provide diversity in vegetative cover types on the landscape.

Response to Fire

Aspen can regenerate from seeds or sprouts. In the West, however, regeneration from seeds is uncommon because the continuously high moisture requirements of young seedlings are seldom met (Schier 1981, McDonough 1979). Spouting predominates. Sprouts or suckers can arise from root crowns but they primarily arise from the numerous lateral roots of aspen (Schier 1976). The development of aspen suckers is largely controlled by the action and interaction of several hormones (Schier 1976, 1981).

Fire interrupts the hormone balance by killing phloem tissue, thus preventing the downward flow of auxin. Also, if the fire is intense enough to destroy aspen foliage, upward movement of cytokinin in the xylem will cease, thus increasing its concentration in the roots.

Sucker Densities

The density of suckers resulting from disturbance depends on many factors (Schier 1976, 1981) including:

1. Density of aspen roots.

2. Health of roots--disease and age influence root vigor. Most suckering comes from newly formed bud primordia produced in healthy roots.

3. Hormone balance (cytokinin-auxin).

4. Carbohydrate reserves--can be a factor in closely repeated treatments.

5. Genotype--expressed in clonal differences.

6. Environmental factors (temperature and light). Temperature affects hormone balance; higher temperatures stimulate cytokinin, but degrade auxin. Aspen develop best in full light.

7. Root depth--deep roots produce fewer suckers because of lower soil temperature and greater carbohydrate needs to get suckers to the surface.

Sucker densities reported from five different fires in northwestern Wyoming (Bartos 1979) and one fire in New Mexico (Patton and Avant 1970) ranged from 20,000 to 150,000/ha during the first few years after fire. On two fire sites sucker densities increased for the first 2 years after fire, then decreased. On the other four sites, sucker densities were highest the first year following fire, then decreased. On all sites, sucker densities appeared to stabilize after 4 years. In mixed aspen-conifer stands, Brown and DeByle (1982) observed 17,000 to 25,000 stems/ha, substantially fewer than reported for the stands dominated by aspen. Study results show that considerable variation in sucker densities can be expected among fires and even within individual burns.

Sucker density appears to relate to fire intensity and fire severity, although information in the literature confuses the effects of fire intensity with fire severity. Fire intensity is the amount of heat released by the fire per unit of time. Fireline intensity is commonly used to describe fire behavior. It is the amount of energy released per unit time per unit length of fire front. Fire severity incorporates both upward and downward heat fluxes and is an expression of the effect of fire on the ecosystem. It relates to the amount of mortality to living plants and other organisms and to the extent of organic matter loss. Fire severity described by Ryan and Noste (in press) has fire intensity and depth of ground char components. The ground char component of severity is often used in reference to fire effects on soil, which involves consumption of the forest floor and downward heat flux. Severe fires may be either of low or high-intensity depending primarily upon moisture contents of the forest floor and soil.

Three levels of fire severity probably relate meaningfully to sucker response based on studies by Horton and Hopkins (1965), and Bartos and Mueggler (1981). They are suggested as:

1. Low fire intensity and low ground char.

   Vegetation is partially killed, no more than one-half to two-thirds of the aspen are killed, litter and upper duff only are consumed. Response: suckering is patchy and relatively sparse, least effective response to fire.

2. Moderate to high fire intensity, moderate ground char.

   All or nearly all aspen are killed, patches of duff and charred material remain. Response: suckering is prolific; most effective response to fire.

3. Moderate to high fire intensity, high ground char.

   All aspen are killed, forest floor is reduced to ash and mineral soil is exposed. Response: suckering is substantial; intermediate between low and moderate severity levels.

Burn severity, reflected by consumption of litter, can influence the depth of roots that produce suckers as observed by Schier and Campbell (1978) in Wyoming. Depth of parent roots producing suckers ranged from 0 to 28 cm. Average depth was greatest where essentially all litter was consumed as shown in the following tabulation:

Litter consumed	Average sucker depth
(percent)	(cm)
0-10	6.0
10-90	7.3
90-100	10.7

A possible explanation is that suckers arose from deeper roots where depth of ground char was greater because roots near the surface were killed by heat. It is also possible that a warmer soil surface where insulating litter was removed stimulated deeper suckering.

Understory Response

Increased productivity of herbaceous vegetation has been observed following fire in aspen forests (Bartos and Mueggler 1981). Production of herbaceous vegetation typically increases for a few to many years following fires of low to moderate severity in perhaps all western vegetation types. The amount of increase and duration depend on many factors besides fire such as on-site and off-site plant composition, moisture patterns, and animal use. Changes in the relative amounts of forbs and grasses and perennials and annuals can be substantial. Some perennials may require years to reestablish their former coverage. Generalization about response of specific species is risky because many factors may influence the response. Total productivity, however, generally increases the first or second year following fire. Shrubs usually respond favorably to fire by sprouting and establishing from seed. By 3 to 5 years after fire, a shrub layer is usually apparent if a source of sprouts or seeds was available.

Grazing Impacts

Responses to fire cannot be discussed without considering the impacts of grazing. Postburn vegetation is especially attractive to big game animals and livestock; thus, the opportunity for animal damage is great. Damage, including mortality, to aspen sprouts and other vegetation is directly related to grazing intensity. Tew (1981) reported that a stand of aspen sprouts can be destroyed by 3 successive years of browsing. Sheep are more destructive than cattle. Sprout height regulates the amount of damage by livestock. After sprouts reach 110 cm in height, damage by sheep ceases to be a problem. This requires 3 years of growth on most aspen sites. For cattle, 4 or 5 years are required for suckers to grow out of reach (Sampson 1919). Light browsing of new aspen sprouts can be tolerated because lateral shoots can develop in place of occasional decapitated terminal shoots.

Big game, particularly elk concentrated on wintering grounds, can cause extensive damage to aspen sprouts and overcome the benefits of fire disturbance in aspen (Gruell and Loope 1974). Small burns are especially vulnerable to damage because big game animals concentrate their browsing on small areas. Where excessive browsing by big game is expected, the best solution for regenerating aspen is to burn large areas. Burning a number of smaller units the same year within a nearby area may be one approach to dispersing animal impacts. Single, large burns that create a mosaic of vegetation are another approach. Temporary reduction of elk herds by hunting may also allow new aspen stands to regenerate (DeByle 1979).

Applying Prescribed Fire

The process of developing fire prescriptions should start with clear land management and fire objectives. Land management objectives should be based on appearance of the landscape years after the fire. Fire objectives should specify what fire itself should accomplish. This process, described in more detail by Brown (in press), is completed with prescriptions of when and how to burn a specific area.

Setting Objectives

Where fire is an effective means to improve range, wildlife, and watershed resource values of the aspen ecosystem, the land management objective should be to maintain the aspen cover type, preferably with a mix of age/size classes. When fire is used to reduce slash and stimulate understory production in conjunction with tree harvesting, the land management objective might be to develop another commercial stand of aspen. If so, achieving some minimum number of aspen stems per acre might be an objective. When the land management objective is to maintain the aspen cover type, however, number of stems per hectare is of minor concern because range, wildlife, and watershed values can be enhanced over a wide range of aspen stand densities.

A reasonable expectation after fire in aspen is variable sucker densities throughout burned areas. If flammability varies considerably within burn units, a mosaic of burned and unburned patches may result. As long as fire of adequate intensity reaches most areas desired for treatment, a mosaic of burned and unburned areas should successfully maintain aspen in a diversity of age classes.

Fire objectives should state what fire can directly accomplish. Basically this involves specifying the vegetation to be killed and the organic matter to be consumed. To achieve effective suckering, the objective should be to kill at least 80% of the aspen in a stand. A rule-of-thumb based on my observations of prescribed fires is that fire with flame lengths averaging at least 0.4 m (1.3 ft) is required to consistently kill aspen. This also is about the minimum flame size for sustained spread of fire in aspen fuels.

In mixed stands, it may be unnecessary to kill patches of pure aspen if surrounding conifers can be removed by fire or harvesting. Regardless of whether fire carries through patches of aspen, reduction of conifer cover may be the primary objective for treating mixed conifer-aspen stands.

Appraising Flammability

Although the aspen forest is generally considered difficult to burn, fuels and flammability vary considerably within the aspen and mixed aspen-conifer overstory types. Fuel components affecting flammability include leaf litter, downed woody material, herbaceous vegetation, shrubs, and conifer reproduction. Quantities of these fuels relate to overstory species, understory community types, grazing, and history of stand mortality. Season of year strongly influences fuel moisture conditions.

In planning prescribed fires in aspen, it is important to choose locations where fuels are sufficiently flammable to offer a high chance of successfully meeting objectives. A fuel classification is being developed to help managers appraise fuels for choosing and planning prescribed fires in aspen and mixed aspen-conifer types. Parts of the classification are described here and flammability factors are discussed.

Five fuel classes were identified that differ substantially in vegetation and potential fire behavior (Table 1). The classification of understories is keyed to amount of shrubs and productivity of herbaceous vegetation. Tall forbs dominate the forb component of high-productivity herbaceous classes and low forbs dominate the forb component of low-productivity classes. Aspen dominates the overstory in three classes: aspen-shrub, aspen-tall forb, and aspen-low forb. Productivity and fuel loadings of herbaceous vegetation are greater in the tall forb than in the low forb group. Conifers dominate the overstory in two classes: mixed-shrub and mixed-forb. Herbaceous vegetation under the mixed overstories was considerably less varied than under aspen, thus one forb group appeared reasonable for classifying flammability.

The fuel classification was initially formed by grouping community types in the Bridger-Teton National Forest (Youngblood and Mueggler 1981) and Targhee and Caribou National Forests (Mueggler and Campbell 1982) based on expected differences in flammability. Live and dead fuels were then sampled in 35 stands representing the initial fuel classes. Fire behavior was predicted using these fuel data as well as a range of wind speeds and fuel moisture contents as input to Rothermel's (1972) fire spread model. The fuel and fire behavior information was evaluated and adjustments made in the initial classification. Primarily, initially recognized tall shrub and low shrub groups were consolidated because they overlapped considerably in fuels and flammability. Although this classification is based on community types found in southeastern Idaho and western Wyoming, community type descriptions for other areas such as the Bear Lodge Mountains and Black Hills (Severson and Thilenius 1976) could probably be evaluated to fit it.

Table 1.--A vegetation classification of aspen fuels and flammability.

	Vegetation - fuel classes				
Characteristics	Aspen - shrub	Aspen - tall forb	Aspen - low forb	Mixed - shrub	Mixed - forb
Overstory species occupying 50% or more of canopy	Aspen	Aspen	Aspen	Conifers	Conifers
Shrub coverage, percent	Greater than 30	Less than 30	Less than 30	Greater than 30	Less than 30
Community type understory indicator species	*Prunus* *Amelanchier* *Sherpherdia* *Symphoricarpos* *Artemisia* *Juniperus* *Pachistima*	*Bromus* *Heracleum* *Ligusticum* *Spiraea* *Calamagrostis* *Rudbeckia* *Wyethia*	*Ranunculus* *Berberis* *Arnica* *Astragalus* *Thalictrum* *Geranium* *Poa*	*Prunus* *Sherpherdia* *Spiraea* *Amelanchier* *Symphoricarpos*	*Ligusticum* *Pedicularis* *Berberis* *Arnica* *Calamagrostis* *Thalictrum*

A summary of fuel data from the sampled stands (Table 2) illustrates several important differences and similarities among the vegetation-fuel classes:

1. Shrubs are a significant contributor to fine fuel loadings as shown by the comparisons of fine fuel loading and shrub coverage.

2. Fine fuel loadings differ substantially among the shrub, tall forb, and low forb classes.

3. Production of herbaceous vegetation in the tall forb class is two to four times greater than in the other classes.

4. Litter loadings differ greatly among individual stands, but the average difference among classes is small and not meaningful.

5. Loadings of downed woody fuel 0 to 2.5 cm and 2.5 to 7.6 cm in diameter also vary substantially from stand to stand. The mixed types appear to have slightly more downed woody fuel than the other classes, which is plausible because conifer crowns produce more small twigs and branches than aspen. Generally, however, average differences among classes appear unimportant.

6. Differences in dead fuel loadings between the aspen-shrub and mixed-shrub types are small. However, they should be regarded separately because conifers in the mixed type are likely to torch out, thus creating a more intense fire. Similarly, the aspen-low forb and mixed-low forb should be classified separately, even though differences in dead fuel loadings are reasonably small. Response of aspen to fire could differ greatly between aspen and mixed classes depending on mortality to conifers and amount of residuel aspen.

Modeled fireline intensity (Fig. 2) supports the generalities based on Table 2 data. The aspen-shrub class is the only class that shows a large enough fireline intensity to support a sustainabie fire at a midflame wind speed of 3 km/h. The aspen-low forb and mixed-forb classes have very low fire potentials in their surface fuels. To help visualize fireline intensity, 35 kw/m (10 Btu/ft/s) corresponds to a flame length of 0.4 m according to Byram's (1959) commonly used equation:

$$L = 0.0775(I)^{0.46}$$

where L = flame length, m
I = fireline intensity, kw/m

Table 2.--Average fuel loadings and shrub cover from sampled stands representing the aspen fuel classes. Ranges in values are in parentheses.

Fuel	Aspen-shrub	Aspen-tall forb	Aspen-low forb	Mixed-shrub	Mixed-forb
			kg/ha		
Herbaceous	750 (260 to 1120)	1490 (1150 to 2260)	340 (200 to 520)	100 (90 to 101)	320 (11 to 620)
Shrubs[1]	3550 (1000 to 6890)	120 (0 to 490)	290 (0 to 710)	3410 (2780 to 4050)	710 (110 to 1510)
Litter	2030 (470 to 3150)	1790 (885 to 2510)	1510 (190 to 3070)	2220 (2150 to 2290)	1880 (830 to 2870)
Fines[2]	6880 (4520 to 10520)	3550 (2210 to 4470)	2720 (1840 to 3730)	6780 (6560 to 7000)	3440 (2410 to 3990)
Downed woody 0 to 2.5 cm	2730 (800 to 4730)	1210 (690 to 1610)	2910 (1640 to 4140)	4750 (3810 to 5690)	3040 (1610 to 4370)
Downed woody 0 to 7.6 cm	7870 (4010 to 14020)	8230 (1690 to 18160)	6410 (3690 to 8520)	7810 (6220 to 9400)	8750 (4580 to 13730)
			percent		
Shrub cover	40 (30 - 60)	10 (0 - 20)	10 (0 - 30)	60 (60 - 70)	20 (10 - 30)

[1] Shrubs include foliage and stemwood less than 2 cm in diameter.
[2] Fines include herbaceous plants, shrubs, litter, and 0 to 0.6 cm downed woody fuel.

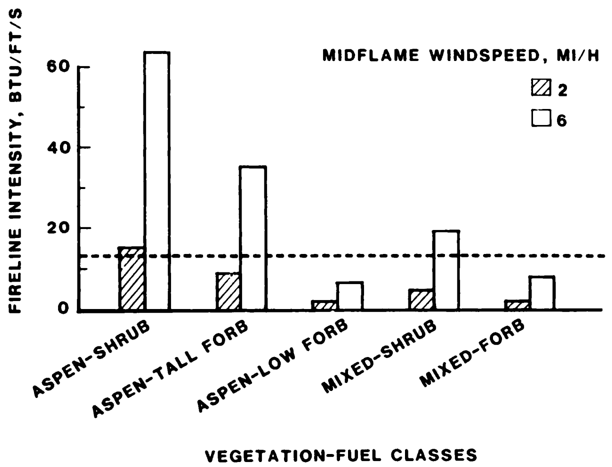

Fig. 2. Fireline intensity modeled under the assumption that 50% of the herbaceous vegetation is cured. Dead fuel moisture content (1- and 10-hour timelag moisture classes) was 8% and live herbaceous moisture was 200%. Sustained fire spread is unlikely below the dashed line.

The probability of successfully using prescribed fire was rated for this classification. A successful fire was defined as having sustained spread and sufficient heat to kill aspen up to 30 cm d.b.h. Probabilities of attaining success were defined on the basis of judgment as:

Good - Loading of downed dead woody and herbaceous fuel adequate for sustained fire spread; good fine fuel continuity. Windspeed and fine fuel moistures come into prescription almost annually.

Fair - Fine fuels mostly from herbaceous vegetation; loadings marginal for sustained spread; fuel continuity is broken and compactness open. Windspeed and fine fuel moisture come into prescription every few years.

Poor - Fine fuels are insufficient to support fire spread. Adequate windspeed and fine fuel moistures to sustain spread occur infrequently.

The probability-of-success ratings in Table 3 reflect the influence of grazing and quantities of downed woody fuel. In planning prescribed fires, categories rated as poor success suggest situations that should be avoided because a favorable payoff is unlikely. Other means of disturbance, particularly cutting, should be used if possible. Efforts to use prescribed fire in aspen should focus on situations having fair to good probabilities of success.

Flammability of the vegetation-fuel classes can be modified substantially by factors that basically involve fuel quantity, fuel moisture content, fuel continuity, and terrain features. Planning of prescribed fires for specific sites must include evaluation of these factors.

Slope.--Slopes less than 30% exert only minor influence on rate of spread and fireline intensity. Slopes exceeding 45% can increase

Table 3.--Probabilities of successfully applying prescribed fire in aspen forests according to vegetation-fuel classes and the influence of grazing and quantities of downed woody material.

	Vegetation - fuel class				
Condition	Aspen - shrub	Aspen - tall forb	Aspen - low forb	Mixed - shrub	Mixed - forb
Ungrazed, light downed woody	good	fair	poor	good	fair
Ungrazed, heavy downed woody	good	fair	poor	good	good
Grazed, light downed woody	fair	poor	poor	fair	fair
Grazed, heavy downed woody	good	poor	poor	good	fair

rate of spread by several times and greatly increase the probability of success. Thus, steep slopes can overcome some deficiency in fuel loadings to support sustained fire spread.

Grazing.--Grazing has the greatest impacts in the aspen-shrub and aspen-tall forb classes because large amounts of herbaceous vegetation can be removed. Measurements of grazing on two sites in these classes showed a two-thirds reduction in herbaceous vegetation. Modeled fireline intensities were reduced 8 to 9 times, indicating sustained fire spread would be impossible. Grazing in the aspen-low forb and mixed-forb classes further reduces an already poor chance of burning success. Generally, to accumulate fine fuels grazing should be deferred during the year before conducting prescribed fires.

Curing.--Curing is the maturation of vegetation through the growing season. Moisture content of vegetation decreases as curing progresses. Once plant tissues are dead, moisture content fluctuates according to atmospheric drying conditions. Curing has a pronounced effect on potential fireline intensity in the aspen-tall forb class and a notable effect in the aspen-shrub class (Fig. 3). I have observed that 50 to 60% of the herbaceous vegetation in these classes should be cured to achieve sustained fire spread with 8 to 16 km/h winds overhead. Curing in the mixed-shrub and mixed-forb types exerts a relatively small influence on fire intensity (Fig. 3). This suggests that prescribed fires might be successfully conducted in this class during late August when temperatures are high and dead fuel moisture contents are low. Waiting until late September or October will result in more curing, but will be offset by higher dead fuel moisture contents.

Quantities of large downed woody fuel.--The contribution of downed woody fuel greater than 7.6 cm in diameter to flammability in aspen cannot be quantified satisfactorily. Nonetheless, large downed woody fuels can increase the chance of successful prescribed fire. Judgment is required to evaluate the importance of large downed woody fuel in meeting fire objectives. Large woody fuels contribute to fire intensity, are a source of ignition for torching conifers, and provide heat to generate indrafts that assist the spread of fire.

Crown closure.--Stands having open canopies are more flammable than those with closed canopies because understory grasses and forbs cure 2 to 4 weeks sooner and wind speeds at the surface are greater due to reduced wind resistance.

Pocket gophers.--Large populations of pocket gophers reduce the chance of successful prescribed fire because they create considerable mineral soil that breaks surface fuel continuity. High wind speeds may be necessary to overcome the fuel discontinuities caused by pocket gophers.

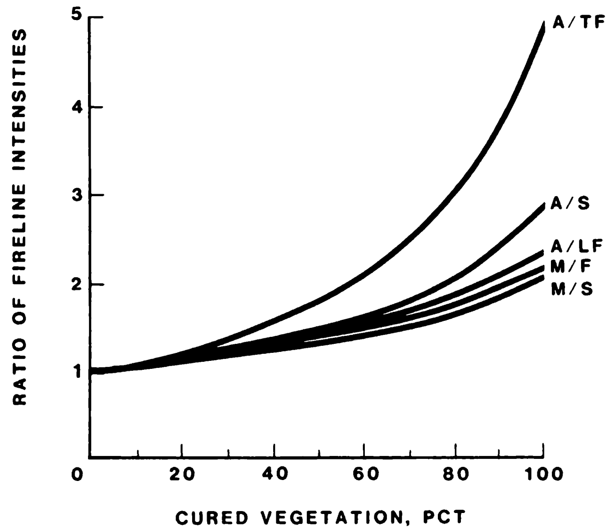

Fig. 3. Ratios of modeled fireline intensities influenced by percent of herbaceous vegetation that is cured. The intensity ratios are calculated as the intensity at any curing level divided by the intensity when curing is 0%.

Curing of Vegetation

A capability to predict live fuel moisture contents would greatly help in determining when aspen can be burned successfully. To explore the feasibility of predicting live fuel moistures using weather indices, moisture contents of perennial grasses and forbs in western Wyoming aspen stands were sampled weekly for 2 years. Seasonal curing trends were similar even though one season was considerably wetter throughout the summer and fall (Brown et. al., unpublished data). Correlations of live fuel moistures with the National Fire Danger Rating System model of live fuel moistures (Burgan 1979) and with the Keetch-Byram Drought Index (Keetch and Byram 1968) were poor and unsuitable as aids for planning prescribed fires.

The best off-site indicator of live fuel moisture contents appears to be simply time of year. Curing progresses steadily through the summer. By early September moisture contents of some plants may have lowered enough to permit fire spread. At this time, on-site evaluation of curing is needed to judge readiness to burn.

On-site evaluation requires an estimate of the proportion of herbaceous vegetation that is cured. Moisture content of cured vegetation responds to rainfall and atmospheric conditions as for dead fuel. Moisture contents of green and transition stage vegetation remain much higher than in the cured stage.

The difference in moisture content between green and transition stage vegetation in this study was relatively small, especially for forbs (Fig. 4). Thus, moisture contents of the green and transition stages can be considered the same for purposes of estimating curing and judging flammability. The transition stage typically is characterized by yellowing of plant parts. Cured leaf tissue shows brown coloration rather than yellow. Cured grass stalks remain straw-colored, but the yellow is largely washed out.

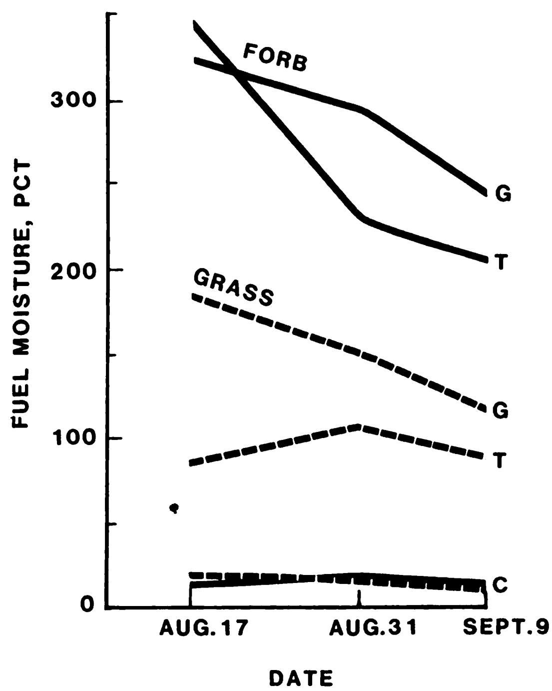

Fig. 4. Moisture contents of forbs (Wyethia amplexicaulis and Rudbeckia occidentalis) and grasses (Bromus and Elymus sp.) on three dates in 1982. The plant tissues were separated into green (G), transition (T), and cured (C) components based on color and texture.

Some other observations from this study (Brown and DeByle 1982) may be of interest:

1. Grasses (primarily Bromus and Elymus) had substantially lower moisture content and cured at faster rates than forbs.

2. Forb moisture contents remained very high, then dropped substantially over a 3-week period. The date of rapid moisture decline varied by species.

3. Grasses and forbs cured much more rapidly in forest openings and under sparse tree canopies.

4. Moisture content of shrub foliage and stems varied little throughout August and September. Moisture content of shrub leaves usually remains high until an abscission layer is formed and color change is pronounced.

Determining when fuels are ready to burn in aspen forests is tricky compared to using prescribed fire in most other fuel/vegetation complexes. Finding the proper time for ignition requires waiting until live fuels are adequately cured, and selecting the time when wind speed and dead fuel moistures are in prescription. Adequate curing is particularly important where herbaceous vegetation is the primary fine fuel, such as in the aspen-tall forb class. Where herbaceous vegetation is a minor component of the fine fuels, little curing may be necessary depending on the quantity of fine dead fuels available for burning. The tradeoff, however, between waiting for further curing to increase flammability and rains that end the burning season means that aspen stands should be burned as soon as possible. Delays in burning will result in fewer accomplishments because the time in prescription is usually short.

A hard freeze can cure live vegetation quickly. Temperatures less than -7 to -10°C can cure forbs and shrub foliage in just a few days. If the freeze occurs before the abscission layer forms, the shrub leaves will remain attached to stems. This adds to the flammability of surface fuels.

Temperatures in aspen stands on slopes are moderate compared to temperatures experienced in openings and valley bottoms. Thus, frost damage under aspen canopies occurs later than in open areas and in low lying areas where cold air collects. Curing assisted by frost under aspen canopies is also delayed. In judging flammability, it is essential to view vegetation inside aspen stands, not just along the edges.

To estimate cured vegetation, walk to a number of points in the area to be burned and estimate the proportion of vegetation that is cured and that is green or in transition. If necessary, adjust the estimates so they total to 100%. We have observed a tendency to overestimate the curing stage that is most abundant by about 10%. Look at the entire plants, particularly near the ground. Grass appears deceptively cured at a distance because seed stalks cure before leaf blades.

In autumn, after most of the herbaceous vegetation is cured, a problem can arise. After early autumn rains, cool season grasses begin growth and greenup occurs near the ground. If the lower 5 cm or more of grass greens up, sustained fire spread becomes difficult.

Fire has been an integral part of aspen forests and has been particularly important in maintaining seral aspen. Prescribed fire can be used to help maintain aspen communities. A successful burning program in aspen, however, will require skillful selection of good burning opportunities and diligence in carrying out prescribed fire plans.

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Impact of Fire on Nongame Wildlife

Edward E. Starkey

Abstract

Fire apparently has a highly variable impact on wildlife. However, the most significant impact is the modification of habitat rather than direct mortality caused by fire. Wildlife species composition and population densities are often relatively unaffected by prescribed burning, but high intensity wildfires can be detrimental, especially to sedentary species or those with narrow habitat requirements.

Both game and nongame wildlife species will benefit from fire management programs which include both prescribed burning and suppression of potentially catastrophic wildfires. Prescribed burning will not only benefit wildlife, but will reduce fuel accumulations and therefore the risk of uncontrolled wildfire.

Introduction

From an ecological perspective, classification of wildlife into nongame and game species is an artificial dichotomy. A species' response to fire and associated habitat changes is primarily dependent upon its evolutionary history, and not its relative value as game. Thus, my primary objective is to review and summarize the existing knowledge of the influence of rangeland fire on wildlife in general, with special reference to nongame species.

Historical Influence of Fire

On many western rangelands fire historically played an important role in shaping and maintaining natural plant and animal communities. Primeval fire frequencies undoubtedly varied among locations and years, but fire was common (Shinn 1980). Therefore species of wildlife occupying these areas were under selective pressure to adapt to the periodic occurrence of fire, as well as changes in habitat caused by burning. In areas of relatively high fire frequencies, species were favored which could successfully utilize early seral plant communities. Thus, fire has been a natural ecological influence, rather than a disruptive force.

With the arrival of European man, fire frequencies were greatly altered. Much promiscuous and abusive burning occurred, followed by a period of almost total fire suppression which in some areas continues. Plant and animal communities have been profoundly influenced. Conversion of large areas of sagebrush-grass to juniper woodlands is a conspicuous example. Wildlife communities have undergone parallel changes.

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In addition to changes in community structure, fuel loadings have been greatly altered. Fire suppression has resulted in an accumulation of fuels in some areas, while fine fuels have been reduced by domestic livestock grazing in other areas. Therefore contemporary fires may differ greatly from primeval fires with regard to intensity, size, and season of burning. Impacts on wildlife have also been modified, by these fundamental changes in fuels and plant communities as well as the implementation of fire management and prescribed burning programs.

Direct Impacts of Fire

When confronted with fire, an animal may escape from the area or seek shelter such as in an underground burrow. The success of either of these strategies depends upon a number of variables including relative size and mobility of the wildlife species, and the rate of spread, size and intensity of the fire. Unfortunately, there has been relatively little research conducted to evaluate the direct impact of fire on wildlife, contrary to the impression created by "Smokey the Bear" advertising campaigns.

However, various studies have suggested that significant numbers of small mammals survive (Gashwiler 1959, Lawrence 1966, Tevis 1956). Using marked individuals, Frenzel (1979) found that mortality of *Peromyscus maniculatus* caused by prescribed burning of sagebrush-grass communities was insignificant.

Lawrence (1966) concluded that small animals survived chaparral fire by occupying unburned islands of habitat or crevices in rock outcroppings. Temperature measurements below the surface of the ground also suggested that animals would be protected from fire by occupying a burrow which is at least three inches below the soil surface, although underground burrows would provide less protection during periods of high humidity. Small mammals survived temperatures above 142°F at 22% relative humidity (Howard et al. 1959) but the lethal temperature may drop as low as 120°F when the relative humidity is above 60% (Lawrence 1966).

Komarek (1969) suggested that wildlife mortalities caused by fire are unusual, even in very large fires observed in Africa. Kern (1981) found that small mammal numbers were reduced immediately after a fire but suggested that the cause was reduced cover and emigration rather than direct mortality from fire.

Where fuel loadings have been greatly increased by long periods of suppression, animals may not be able to escape or avoid catastrophic fires. Significant numbers of animal kills have been reported for chaparral wildfires in California (Chew et al. 1959).

Quinn (1969) found that all species of small mammal except the kangaroo rat (Dipodomys heermanni) were eliminated from a burned area by an intense fire. Kangaroo rats apparently remained in their burrows and were protected from heat.

Not all wildlife species avoid or attempt to escape from fire. Komarek (1969) observed 24 species of birds which seemed to be attracted to fire and smoke. These included hawks, shrikes, and swallows feeding on prey displaced by burning.

Impacts of Habitat Modification

Changes in Species Composition

The primary impact of fire on population densities of wildlife is the alteration of habitat (Wright 1974). Early successional wildlife species are favored by fire, while those associated with climax communities may be eliminated or displaced.

However, Bendell (1974) reviewed a number of studies and found that most species of birds and mammals remained after forest fires. He suggested that this may be caused by uneven burning which leaves a diversity of habitats, and a wide tolerance to disturbance by the species involved. Following fire, new bird species were found in burned areas, but this was not the case for mammals. Apparently the high mobility of birds allowed them to colonize new areas readily.

Frenzel (1979) found no significant change in the number of small mammal species occupying a burned area of sagebrush-grass one year after a fire. Quinn (1969) observed a significant short-term decrease in species of small mammals following fire, but stated that neither wildfires nor prescribed fires extirpate small mammals from chaparral. However, small mammal species with rather narrow ecological niches, such as shrews, may not be able to sustain populations if patches of unburned vegetation do not remain following a fire (McGee 1982).

Species diversity of birds may be increased by prescribed burning in shrub-steppe plant communities, but high intensity wildfires can decrease diversity (Tiagwad et al. 1982). Prescribed burning commonly results in a mosaic of burned and unburned vegetation and therefore increases habitat diversity, which is reflected in the species composition and diversity of associated bird communities. In addition, avian species diversity is related to foliage height diversity (MacArthur and MacArthur 1961), which can be greatly reduced by a high intensity fire. Thus Tiagwad et al. found that following wildfire, avian diversity values were low and most breeding birds were ground nesters.

Changes in Abundance

Bendell (1974) states that bird and mammal densities change very little following fire. He further suggests that this population stability results from the predominant importance of intrinsic mechanisms of population regulation, rather than external factors such as habitat disturbance. This suggestion is supported by the findings of Frenzel (1978), that prescribed fire caused no significant change in population densities of small mammals inhabiting sagebrush-grasslands.

The relative abundance of various species may be influenced, however. McGee (1982) found that total small mammal numbers were not depleted by fire, but species showed a differential response to fire. Deer mice (Peromyscus maniculatus) and Uinta ground squirrel (Spermophilus armatus) populations increased after both spring and fall prescribed burns. He contended that this reflected the relatively broad ecological tolerance of these two species.

Lawrence (1966) found that populations of small mammals and brush-dwelling birds declined rapidly following a chaparral fire. Similarly, rodent populations decreased by 23% to 91% following burning of veld in South Africa (Kern 1981). Bird densities also decreased significantly following wildfire in shrub-steppe of northern California, although densities were increased by prescribed fire (Tiagwad et al. 1982).

Difficulties in Predicting Fire Impacts

Conflicting research findings can often be explained by differences in methods or techniques, and this is likely the case for many of the differing research results cited above (Bendell 1974). However, research on the impacts of fire on wildlife is greatly hindered by a lack of quantitative and descriptive information on specific fires of concern. Frequently, published reports concerning the influence of fire on wildlife contain little or no information on such important variables as temperature, humidity, wind velocity or fuel moisture at the time of the burn. Also, even for prescribed burns, pre-burn descriptions of fuels and vegetation are frequently inadequate.

Thus, wildlife researchers must work closely with fire scientists and obtain a basic understanding of fire behavior and impacts on plant communities. We must understand that not all fires are alike, and that high intensity wildfires have different impacts than most prescribed or controlled burns.

In addition, long-term studies of wildlife populations and fire must be conducted. Most published reports describe wildlife response to burns during only the first few years following fire. Short- and long-term impacts of fire may be very different. For example, Klein (1982) has suggested that fire may be destructive of caribou forage (lichens) on the short-term (less than 50 years), but over long periods (greater than 100 years) essential for maintenance of ecological diversity and forage production.

Conclusions

Fire apparently has a highly variable impact on wildlife. Numerous studies provide conflicting evidence concerning wildlife response to fire, however, the following conclusions seem reasonable:

1. Fire usually does not cause significant direct mortalities, most animals are able to hide or escape.

2. The most significant impact of fire is the modification of habitat, by reduction of cover and changes in plant communities.

3. Although burning does result in some changes in wildlife species composition, many species are relatively unaffected.

4. Population densities are also relatively stable following fire, although some studies have reported significant declines in abundance of both small mammals and birds.

5. Differences in variables such as weather, fuels and plant communities may account for conflicting results of various studies.

6. Research on impacts of fire on wildlife must include adequate descriptive and quantitative information on specific burns, with an increased emphasis on long-term studies.

From a management perspective it is important to differentiate between the impacts of wild and controlled fires. Historically, rangeland fire was common and wildlife evolved under its influence. However, wildlife species were not usually exposed to catastrophic fires burning in heavy fuels. Thus uncontrolled fires such as those common in chaparral areas may significantly reduce nongame wildlife populations for a few years. Although prescribed fires do not exactly simulate primeval fires, wildlife diversity and abundance are frequently increased following prescribed burning.

Thus, both game and nongame wildlife values will be enhanced by a fire management program which includes both prescribed burning and suppression of potentially catastrophic wildfires. Prescribed burning will not only benefit wildlife, but will reduce fuel accumulations and therefore the risk of uncontrolled wildfire (Martin et al. 1976). Optimum benefits for wildlife will result from a prescribed burning program which mimics historic fire frequencies and sizes, and creates a mosaic of different aged successional stages (McGee 1982).

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Sage Grouse Life History and Habitat Management

Robert Autenrieth

Abstract

The perpetuation of sage grouse (Centrocercus urophasianus) in Idaho and the Northwest may be equated to the success or failure of land management agencies in promoting habitat diversity. Only eight states have harvestable populations and in Alberta, Canada hunting is by permit only. While not a threatened or endangered species, reduced sage grouse numbers and their elimination from much of their historic habitat directly relate to vegetative conversions for agriculture and grazing.

Where annual precipitation is above 30 cm, the range has often endured or recovered from past abuses and sage grouse numbers are relatively high. However, most xeric ranges (below 30 cm) have not recovered and sage grouse numbers tend to be low. It is on these ranges that future "range improvements" will have the greatest impacts.

In order to address the impacts of land management practices on sage grouse, an understanding of how habitat requirements change during the annual cycle is essential:

1.) The lek is a key area because it receives traditional use and provides wildlife managers trend data for overwinter survival. It is also the hub from which nesting occurs. The lek itself should not be the focus of habitat managers concern because it is actually nothing more than an open space.

2.) The quality of nesting habitat surrounding the lek is the single most important factor in population success. Where a 35% sagebrush (Artemisia spp) canopy and 60 cm height are combined with residual grass cover, the probability of predator detection is significantly reduced. The percentage of successfully nesting hens and the juveniles/100 adult females ratios are significantly higher in areas with robust shrub-grass production and where forbs are a common component of the spring range.

3.) Broods require forbs, insects and cover for growth, concealment and shade. Where these requirements are met at or near the nest site, the brood moves less, reducing exposure to predation and conserving energy. On dry ranges, however, broods are forced to move to the nearest meadow for attaining their needs. Under these conditions, competition with livestock is the most significant. Although the water available at meadows is the important commodity for livestock, the associated consumption and trampling of forbs reduces their availability for sage grouse broods. Providing stock water outside fenced meadows could improve brood success on many xeric ranges.

4.) Sage grouse consume mostly sagebrush in the winter and also use it as thermal cover. Their migration to traditional winter ranges relates to snow depth. During the drought winter of 1977, when little snow accumulated in southern Idaho, sage grouse were widely distributed. During recent severe winters, sage grouse were documented in large groups on low elevation ranges where they had not previously been observed. In some areas, agriculture has removed the sagebrush on these historic winter ranges used only during severe conditions. Winter ranges are often proposed for sagebrush eradication to improve livestock range because of their relatively high canopy coverage. No decisions on sagebrush control projects should be made without documenting the percentage of sagebrush still available to wintering sage grouse during maximum snow accumulations.

In summary, optimal sage grouse nesting and wintering habitats require sagebrush height and density not necessary for brood rearing areas. Producing brood rearing habitat using treatments that leave a mosaic pattern of brush can be beneficial for both sedentary and migratory sage grouse populations. Controlled fall burns in areas of 30 cm or greater annual precipitation and where cheat grass (Bromus tectorum) and rabbitbrush (Chrysothamnus spp) invasions are not a problem can enhance brood rearing habitat. Treatments should be conducted in patches that do not exceed 10 to 15 acres. Sagebrush removal seldom benefits sage grouse unless increased forb production occurs.

The challenge for land management agencies is to avoid replacing one monotype with another, and to promote habitat diversity through treating small acreages in mosiac patterns. The future abundance and diversity of wildlife will probably be the best measure of success.

(Abstract only is available for this paper)

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Big Game Response to Fire in Sagebrush-Grass Rangelands

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Abstract

Big game respond to fire in their habitat. Fire can improve production and quality of forage and retard successional advance of seral grassland. It has potential in bighorn sheep, pronghorn antelope and mule deer habitat under proper management. On mule deer shrub climax winter ranges it is possible to have short-term and long-term benefits, but there is a great possibility of both short-term and long-term losses as well. Frequent wildfires have led to losses of winter habitat.

Introduction

Big game respond to fire in their habitat. Like many other wildlife species, they display an affinity for subclimax plant associations and disturbance, such as wildfire, must have been a common occurrence throughout evolutionary time. Substituting prescribed fire for wildfire, therefore, appears to have biological justification.

Role of Fire

A major role of fire in big game habitat is thought to be related to nutrition. Intuitively, we expect increases in forage quality in burned areas and a corresponding positive animal response.

But little research has been done to test this belief. Hobbs and Spowart (1984) in Colorado, reported on one such study on mountain sheep (Ovis canadensis) and mule deer (Odocoileus hemionus) winter and spring range. Their burns in mountain shrub and grassland communities substantially improved the quality of winter diets of both the sheep and the deer.

Both species were able to find more green grass on the burns compared to the controls. Green grass was more nutritious than other forages during the winter and was more available to wildlife on the burns. Old growth obscured the green grass on the unburned plots and the authors believed it may have inhibited feeding by mule deer. Also, the blackened soil surface on the burned plots was thought to favor the grass growth they noted during the winter.

In May, the unburned controls provided diets nutritionally superior to those on the burns. This was the normal green up period and those plants were phenologically younger than the plants on the burns. Green up had occurred 1-2 weeks earlier on the burned plots. The ungulates responded to it. Later the plants on the unburned plots developed, providing a second distinct flush of nutritious plant forage. Burning, therefore, prolonged the period of young forage at a critical time.

In an Idaho study (Peek et al. 1979) found these same wildlife species responded to small prescribed burns on Wyoming big sage (Artemisia tridentata wyomingensis) - bluebunch wheatgrass (Agropyron spicatum) rangeland. The burned plots were grazed more heavily than adjacent ungrazed sites for 4 years after burning. The authors suspected that most winter use on bluebunch wheatgrass was made by bighorn rather than mule deer, but were unable to distinguish the relative amount of use. Bighorns reportedly utilize higher levels of grass than do mule deer.

They believed there are two benefits of fire: (1) fire may improve production and palatability of forage and (2) it may retard successional advance of seral grassland towards conifer climax. They believed they had achieved the first in their study.

The second benefit was demonstated in Nevada. A study was made of burns in singleleaf pinyon (Pinus monophylla) - Utah juniper (Juniperus osteosperma) communities to determine successional changes and mule deer use (Stager 1977). There were eight ages of burns ranging from 2 to 115 years. Pellet accumulations and tracks were used as indicators of mule deer use.

The pattern of succession paralleled the results of other studies in the pinyon-juniper type (Arnold et al. 1964, Barney and Frischknecht 1974). Forbs, particularly annuals, were abundant up to four years following burning. Up through 16 years, there were significantly more forbs than in the unburned sites. One site 24 years old had significantly more forbs, indicating this may be about as long as a significant forb response can be expected. Grasses, on the other hand, appeared to respond later than forbs. The burns age 24, 45 and 115 years had the most grass basal area. Grass improvement was still notable at 115 years.

Shrubs suffered declines during the first several years but had recovered by 16 years and, typically, were more abundant on the burned sites than on the unburned from then through 115 years following fire. On two sites, where 24-year and 45-year old burns were compared with 115-year old, the younger burned areas had a greater canopy cover of shrubs than the older. In general, however, shrub cover was maintained on the 115-year old areas.

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The impact of fire on pinyon and juniper cover was still very evident 115 years following fire.

Mule deer favored the unburned areas over the burned from ages 2 to 4. From 16 years through 115 years, deer found the burns suitable using them equally as much or more than the unburned adjacent areas. One exception was that tracks were less abundant on one 115-year. old burn than the unburned. Generally, the 115-year old burns were still in a successional stage desirable for mule deer.

Species Response

Bighorn Sheep

As implied in earlier paragraphs, fire can be a tool useful for habitat improvement of bighorn sheep on sagebrush-grass rangelands. Bighorns are grassland inhabitants (Geist 1971). While their winter range may be a shrub-grass steppe (Peek et al. 1979), primary forage at that time is grass (Hanson 1982). On a year-long basis, Van Dyke et al. (1983) reported grass was the major diet item (59%) followed by forbs (32%) and shrubs (9%). Also, on the Sheldon National Wildlife Refuge, bighorns consumed 42% grass and 40% forbs year long (Hanson 1982). Both these studies showed grass or forb preference from cold desert sites where shrubs predominated.

Hanson's (1982) recommendations for the Sheldon, considering the needs of domestic cattle (Bos taurus), feral horses (Equus caballus), pronghorn antelope (Antilocapra americana), mule deer and bighorn sheep, was that management be geared to increase both native grasses and forbs. Prescribed burning could play a part, he believed, in reducing, but not eliminating shrubs. With desert bighorns in Nevada and California, it appears that lack of fire has been detrimental to bighorns as shrubs increased in their habitat (Graham 1981). Graf (1981) recommended prescribed burning to reduce pinyon and juniper in desert bighorn habitat.

Pronghorn Antelope

For pronghorn antelope it is suggested that fire can be used on a small scale to enhance their habitat (Kindschy et al. 1982). Fire could improve forage. While from 45% to 54% of their diet was browse in Hanson's (1982) Nevada study, there was nearly an equal proportion of forbs. Kindschy et al. (1982) believed that prescribed burning, used properly, could decrease sagebrush density and encourage grasses and forbs. Fire triggers a response in forbs that enhances the habitat and attracts pronghorns. This attraction draws pronghorns into young burns, using areas they may not have occupied for a number of years prior to the fires.

Mule Deer

Fire generally is considered to be a favorable influence in mule deer habitat. In fact, Wallmo (1978), as an axiom for mule deer habitat management, stated, "Early stages of plant succession are more beneficial than climax vegetation." Greater abundance, diversity and quality of forage is believed to occur in the secondary succession.

This principle has been applied widely to the forest types with results similar to that already presented by Stager (1977) for pinyon-juniper communities.

Another pinyon-juniper study in Arizona (McCulloch 1969) reported favorably for burning. Mule deer responded to the burn, using the burn more than the unburned woodland during most of the fall-winter period. Pellet accumulation rates were high in the burned zone at 400-800 m from live woodland. However, accumulation rates were greater in the burned zone nearest to unburned woodland. After 13-15 years, woody cover on the burned area consisted of numerous dead trees and sparsely scattered clumps of shrubs. Browse plants for deer were more abundant in the unburned woodland but grasses and forbs were more abundant on the burn.

McCulloch (1974) later listed recommendations for modifying pinyon-juniper woodlands for deer and elk. He recommended numerous small clearings 30-200 m in width interspersed in large areas of dense, untreated woodlands. The clearings should be long, narrow strips. Areas already used by deer and elk should be selected for conversions as these areas frequently contain potentially abundant supplies of desirable forage but have limited forage production due to the dense overstory. These guidelines, have their parallel in those developed for forested areas (Thomas et al. 1979), and it would seem forest guidelines often have common application to woodland vegetation.

Where mule deer inhabitat non-woodland sagebrush-grasslands, the role of fire is less clear. The Great Basin has many such areas. The deer may summer in higher elevation forest types, woodlands, aspen or mountain brush areas. The summer ranges are often isolated and if shrub and tree cover is minimal, as it commonly is in the Great Basin, space and topography sometimes substitute for vegetative cover in a fashion described by Leckenby et al. (1982). On these summer ranges whatever cover is available is valuable when it is limited in amount. Under those circumstances, any type conversion of aspen (Populus tremuloides), mountain mahogany (Cercocarpus ledifolius), riparian willows (Salix sp.), or other tall shrubs or trees is viewed as manipulation of critical cover.

Summer ranges are important and critical habitats for these mule deer. In some cases, these ranges are recognized as being the key to healthy deer populations (Pederson and Harper 1978, 1984) and summer range quality may influence herd production as much as winter range quality.

In summer habitat in poor condition, fire can play a role. Recognizing that cover may be critical, foraging areas may benefit from fire. Fire should be used as a tool to enhance forb production. Small burns, interspersed in unburned shrub and tree dominated areas can be beneficial.

On the winter ranges of some of these herds, fire is usually viewed as detrimental. These ranges lack woody cover, often with antelope bitterbrush (Purshia tridentata) and sagebrush providing nearly all the vegetative cover that exists. Hiding and thermal cover is marginal, therefore viewed as critical. Since shrub cover is marginal, isolation and topography usually are necessary features of an important winter range. The ranges are located on the lower elevation areas and many suffer from encroaching urbanization or other human use and activity. This has restricted their acreage, making the remaining ranges more valuable for maintaining mule deer population levels.

Fire on Shrub Climax Winter Ranges

Fire seemingly has little place in this habitat since antelope bitterbrush and sagebrush, both important for food and for cover on winter ranges, are normally eliminated when burned (Wright and Bailey 1982). How can fire be beneficial in this situation?

The interaction between mule deer and fire on these shrub winter ranges is exemplified by Klein (1982) on the relationship between fire, lichens and caribou. In the long term, fire in caribou habitat maintains diversity of vegetation types and rejuvenates old forest stands with declining lichen productivity. On antelope bitterbrush winter ranges, fire may serve the same two purposes, creating diversity and rejuvenating quality and productivity. Old bitterbrush stands burn and new stands develop from rodent cached seed stores. Following the burn, the lack of shrubs has a negative effect on mule deer. The deer avoid the large burned areas. At least, use of the burned areas drops during the winter months, although there may be an attraction to some burns in early spring or even in late winter when unseasonal green vegetation is available.

Judging from observations in northwestern Nevada, from 10 to 15 years following fire, bitterbrush may again be established to the degree that mule deer initiate using the burned areas. This timing varies dependent on distribution of seed caches, climatic conditions that favor establishment and the cattle use of the area. A drouth can lead to the failure of re-establishment just as it can cause a seeding to fail. Intensive use by cattle reduces the rate of bitterbrush re-establishment for bitterbrush is a forage species readily browsed by cattle.

Once bitterbrush and other shrubs are abundant enough to begin attracting mule deer, from 10-15% total shrub canopy cover, the shrubs are young, vigorous and productive. As the burns tend to be large acreages, and mule deer had been avoiding the burn sites, the young plants have had opportunity to grow relatively unaffected by browsing. With further development of the stands, mule deer use increases and stabilizes. Finally, the stands once again reach old age and stagnate. How long this may take, I do not know, possibly 50 to 75 years.

Short-term vs. Long-term Effects of Fire

Just as Klein (1982) found in the relationship between fire and caribou ecology, we have failed to distinguish between short-term and long-term effects of fire in Great Basin mule deer shrub winter ranges. The cycle is much shorter in our mule deer habitat than for caribou. Short-term refers to the immediate reduction in suitability of winter habitat with a drop in deer use that may take 15 years to recover. Long-term effects are those related to the re-establishment of a rejuvenated winter range and the diverse pattern that develops in a landscape with varying-age plant communities as a result of fire. The long-term span in this case extends for 50 to 75 years.

Klein (1982) classified fire effects to exemplify the distinction between short-term and long-term consequences of fire. I have adapted portion of his classification to our situation for mule deer.

Short-Term Effects

Destruction of winter browse forage:

> Degree and duration of effect relates to size, intensity and completeness of burn; previous vegetation types present and their seral stages; and the availability of alternative wintering habitats.

Reduced availability of forage in post-fire areas:

> Little browse is available in recently burned areas. Early successional stage vegetation has value for mule deer as early spring forage but due to the lack of cover, its use tends to be on the edges of burns.

Low intensity burns may improve forage quality:

> Fires of low intensity may release nutrients, stimulate grass and forb growth of

high nutritive value and not kill antelope bitterbrush (Klebenow and Beall 1971). If small in size, they are beneficial.

Long-Term Effects

Maintain diversity in vegetation types:

Fire in association with land form is a major element maintaining plant successional sequences in the Great Basin; the interspersion of young, intermediate and old shrub-grass-forb communities; and productivity of bitterbrush.

Rejuvenate old bitterbrush-sagebrush stands with declining winter forage quality and productivity:

Bitterbrush production declines in old stagnant stands. Understory may be depleted. Fires return land to early successional stages.

Can create extensive monotypes under certain conditions:

Fires may burn vast areas, leading to uniformity of vegetation type, thus creating long-term irregularity in productivity and availability of bitterbrush browse. Mule deer populations respond accordingly.

Replacement of shrublands with grasslands:

Burning may favor the establishment of annuals like cheatgrass (*Bromus tectorum*) which may increase the frequency of fires. Repeated frequent burning favors the establishment of a grassland, delaying the return of the shrubland climax. Grasslands have less carrying capacity for wintering mule deer than shrublands.

Implications for Management

Fire does play a role in big game management of sagebrush-grass rangelands. With bighorn sheep, adapted to a grassland habitat, it is the mechanism that maintains Great Basin habitats at a successional stage most suitable for this species.

It plays a similar role with pronghorn antelope. This big game animal, adapted to a grassland-shrubland habitat, responds to the forb production that results from successionally young communities. Shrubs play an important role in their year-long diets. A fire management program that recognizes and responds to the need for shrubs in pronghorn habitat, can be successful. A management goal should be the development of successional stages that leads to a grassland-shrubland landscape.

Mule deer also respond to fire. In woodland communities, they benefit from the increased quantity and quality of forage produced in the subclimax plant associations created by fires. In the early stages they favor the edges near cover. From 16 years onward the burns have recovered enough to be suitable for deer use and remain that way for a long time.

In climax shrubland communities that comprise mule deer winter ranges, fire often has a negative influence. Wildfire can be catastrophic events that vary in frequency and extent, leading to declines in carrying capacity for wintering deer. Furthermore, cheatgrass invasion and livestock use can modify natural patterns. Wildfire cannot be counted on to burn in patterns, in sequences and to the extent that favor optimal use by wintering deer. Short-term losses of winter ranges may therefore lead to variations in mule deer health, production and population sizes. Where wildfire can be effectively controlled, prescribed burning that rejuvenates decadent antelope bitterbrush stands can be employed as a long-term management tool.

Long-term monitoring of the responses of all big game species and the varied types of vegetation to fire clearly should be a high priority of researchers and managers in big game rangelands.

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Effects of Fire on Rangeland Watersheds

John C. Buckhouse[1]

Abstract

The watershed ramifications of fire are varied and only partially understood. Several axioms are present, however. They are:

1. Fire creates changes in vegetation which in turn have temporary or permanent watershed effects.

2. Grasslands frequently burn with rapid, relatively cool fires. Therefore, grass plant crowns may not be killed and seeds lying next to the soil frequently survive.

3. Fires in forested watersheds may be extremely variable in terms of intensity, temperature and duration. Fire, under certain environmental and fuel conditions, may creep through the litter or it may "crown out"--killing all vegetation and sterilizing the ground and retarding regrowth.

4. Overland flows of water are usually adversely affected immediately following fire. Variable time frames are required for recovery. Soil depth appears to be a key element in regrowth and recovery.

Introduction

The watershed response to fire is varied and complex. A paucity of information is available, and much of what is published has been gleaned serendipitously from wildfires (no two of which are alike) or parenthetically from controlled and prescribed burns.

Watershed management can be defined as the management of all the natural resources of a watershed to protect, maintain, or improve its water yields. Therefore, fire as a manipulative tool has, under appropriate circumstances, a legitimate role in the management of watersheds.

Watershed Manifestations of Fire Within Rangeland Ecosystems

Woodlands

Quantity of Water Yield

The amount of water which may be available for downstream flows as streams or as springs and seeps is variable. A reduction in woody vegetation with its attendant reduction in transpiration by the trees can result in excess soil moisture which enables springs to appear and streams to flow longer or more vigorously. This seems to occur primarily at the upper precipitation limits of the woodland zone. At the lower margins of the zone, destruction of trees and subsequent replacement by herbaceous vegetation may not result in a net transpirational/soil moisture gain.

Timing of Water Yield

Infiltration rates of water crossing the air-soil interface are likely to decline following fire. For example, infiltration rates fell from around 8 cm/hour to about 5 cm/hour following a southeastern Utah controlled burn in a pinyon-juniper woodland.

Depending upon heat intensity and soil texture, infiltration rates may decline to such a degree that "hydrophobic" conditions occur. Commonly quoted temperature ranges suggest that if the soil temperature is less than 300°F the soil will be unaffected by fire and infiltration rates will be essentially unchanged. From somewhere around 300°F through to 900°F, hydrophobic conditions increase and intensify. Finally at even higher temperatures, hydrophobic conditions wane, with infiltration rates resembling those associated with that type of mineral soil. Apparently fire, at certain temperatures, recombines the organic component of the soil creating the hydrophobic phenomenon up to the point that it is totally consumed and the rates revert to those of the mineral soil.

With an increase in bare ground and the possibility of hydrophobic soils, infiltration rates decrease and the possibility of overland flows increases. Consequently, immediately following fire one can expect that storms will result in flashy runoff.

Quality of Water Yield

Sediment production is the major detriment to water quality which one might expect following fire. Of course, the severity of the burn dictates the amount of bare ground exposed, the percentage of over- and understory vegetation kill, and tendency toward hydrophobic condition. If fire conditions are such that grass crowns are not killed and hydrophobic soils are not created, one might expect to experience a rapid recovery with a more complete herbaceous ground cover than before the blaze. This being the case, the area may, within months, be less erosive than when it supported woodland species.

Nutrient release may also occur following fire in woodland ecosystems. Research indicates that increases in phosphorous, potassium and calcium are found in runoff waters following fire. Sodium response is variable.

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In some cases increased sodium was found in runoff waters and in other cases it decreased following fire. Nitrate-nitrogen values in runoff waters from many woodland ecosystems are generally low prior to fire and remain low following fire. It is unclear whether fire ties up or volatilizes the nitrogen, or whether the amount is simply so naturally low that it is difficult to detect.

Whether one considers nutrient release to be positive or negative depends a lot on point of view and the location of the burn in relation to the stream courses. Nutrient release following fire may result in a significant "nutrient cycling", providing a fertilizer for herbaceous vegetation with resultant increases in vegetative cover--a hydrologic benefit. Conversely, if nutrient release results in eutrophication of water supplies rather than upland soil fertilization, a hydrologic debit is realized.

Management Implications

Absence of fire may be much of the reason for invasion of rangelands by woody species. Associated with woodland encroachment on many rangelands are high erosional rates and decreased infiltration rates as the woodland species outcompete herbaceous species. Fire, under proper conditions, may provide the mechanism for improved watershed and multiple resource production.

Shrublands

Quantity of Water Yield

Many shrubland ecosystems occur at relatively low precipitation zones. Therefore, soil moisture deficits are common. It is unlikely that an increased downstream water supply will result as a function of shrub removal by burning. Precipitation which once may have been intercepted by the shrub canopy or transpired by its metabolism generally is effectively consumed by the herbaceous vegetation which replaces the shrubs.

Timing of Water Yield

As the organic component on the soil surface is removed and bare ground is increased, raindrop splash erosion and surface soil puddling occur. As a consequence, infiltration rates decrease and overland flow increases. Shrublands subjected to storms immediately following fire can be expected to exhibit flashy runoff events.

Quality of Water Yield

The quality of the water yields associated with shrublands is directly tied to runoff. Conditions which encourage overland, rather than subsurface, flows encourage the detachment and entrainment of soil particles. The on-site erosion and off-site sedimentation associated with runoff immediately following fire is the major water quality hazard of fire in this ecosystem.

Management Implications

Prescribed burning may be helpful in some instances within this ecosystem. Generally fire is most useful on sites with relatively deep soils and with an established perennial grass understory. Examples may be a north exposure sagebrush/fescue site or a crested wheatgrass planting which has been invaded. Poor places to burn would be desirable shrub sites which are susceptible to fire or undesirable shrub species, such as rabbitbrush, which tend to sprout vigorously following fire. These are poor sites to burn since the potential resource benefits associated with burning may not justify the hydrologic risk undertaken.

Grasslands

Quantity of Water Yield

Burned grassland sites will rarely result in increased downstream flows since transpirational savings by removal of the herbaceous vegetation is minimal.

Timing of Water Yield

Infiltration rates are decreased with removal of the vegetative component of this system. Therefore, one can expect decreased subsurface flows and increased surface runoff following fire and before regrowth occurs. Thus, flashy periods of runoff are to be expected prior to vegetation reestablishment.

Quality of Water Yield

Prior to revegetation of a burned grassland, one can expect increased sediment loading in the runoff.

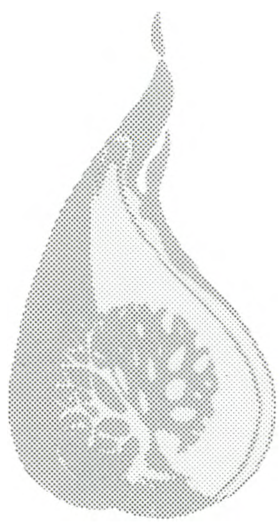
Management Implications

Flashy, turbid runoff can be expected from burned grasslands prior to revegetation. Revegetation may or may not represent much of a problem, however, depending upon the density and species of herbaceous vegetation present prior to the fire. One may wish to burn grasslands in order to rejuvenate stagnate stands, to remove dense litter mats in order to initiate earlier spring growth, or to manipulate grass phenology and palatability for domestic or wild ungulates. It may also be possible to control noxious vegetation in grasslands through burning. However, in general it is difficult to control annual grasses with fire unless one is willing to accept the hydrologic risk of repeated burnings (to deplete the seed reservoir) and its associated repeated periods of post-burn, pre-vegetation establishment vulnerability.

Conclusions

Watershed benefits as well as debits can accrue on any given watershed as a result of fire. Prescribed burns can result in vegetation rejuvenation and/or conversions which are hydrologically positive. It may also result in periods of watershed vulnerability associated with periods of increased bare ground, decreased infiltration rates, accelerated nutrient release, and high erosion potential.

In general, flat ground, deep soil, existing perennial herbaceous vegetation, and cool fires represent the lowest watershed risk conditions for prescribed fire. With proper care and planning, a myriad of multiple resource objectives--with a minimum of hydrologic risk--is possible.



Vegetation Changes Following 2,4-D Application and Fire in a Mountain Big Sagebrush Habitat Type

Larry Mangan and R. Autenrieth

Abstract

Areas in a mountain big sagebrush (Artemisia tridentata vaseyana) zone in southern Idaho that were burned according to prescription were compared to control plots and areas sprayed with the herbicide 2,4-D to detect differences in vegetal response and effects on wildlife. Frequencies of forbs palatable to sage grouse (Centrocercus urophasianus) and elk (Cervus elaphus) were greater in the burned areas than in either sprayed or control plots. Frequencies of most grasses were similar in treated and control plots; however, production of western needlegrass (Stipa occidentalis) increased 5 fold over the control 2 years after an October burn. Fall burns in a mountain big sagebrush zone can benefit sage grouse and elk.

Introduction

Abusive livestock grazing practices have promoted unnaturally dense stands of sagebrush (Artemisia spp.) throughout the West. Decreased production of native grasses and forbs in these dense stands has reduced their forage value to livestock and some wildlife species depending on time and type of use. Land managers have a variety of mechanical and chemical techniques for reducing sagebrush and improving range condition (Plummer et al., 1968). Each technique has its own merits and pitfalls depending on environmental and economic factors and objectives of the project.

The herbicide 2,4-D (2,4-dichlorophenoxy acetic acid) was used extensively in the past to reduce brush and increase grass, primarily for livestock. This chemical selectively kills broad leafed plants, including forbs important to sage grouse (Centrocercus urophasianus) and other wildlife species. The negative impacts of using 2,4-D to remove sagebrush in sage grouse habitat have been documented (Braun et al., 1977). However, few data are available regarding prescribed burn impacts on wildlife. Clifton (1981), found no increase in forbs during 2 years following a fall burn in a Wyoming big sagebrush (Artemisia tridentata ssp. wyomingensis) site in Idaho and found no effect on sage grouse. Two more recent studies in southeast Idaho showed that a relatively small burn (4.5 ha) only benefitted sage grouse and antelope (Antilocapra americana) through increased forb production and/or a lengthened season of forb growth in the spring (Gates 1983).

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Martin (1985) found that in the Dubois area, wildfires did not decrease attendance on leks located within the burn perimeters. The "Sage Grouse Management Practices" bulletin (Western States Sage Grouse Committee, 1982) indicates late fall burns may benefit brood rearing habitat where it is limited but burning should seldom be conducted on winter or nesting areas.

In this paper we examine vegetative responses following prescribed burns and 2,4-D spraying in habitat important to sage grouse, elk (Cervus elaphus), mule deer (Odocoileus hemionus) and livestock. We conclude that prescribed burning in the mountain big sagebrush (Artemisia tridentata vaseyana) zone can improve conditions for these species by increasing the availability of key forbs and grasses.

Materials and Methods

Our study area was on the north end of the Bennett Hills, 32 km southeast of Fairfield, Camas County, Idaho. Treatment areas ranged in elevation from 1530 to 1650 meters. Precipitation at Fairfield averages 36 cm with 89 percent usually falling between October and June, largely in the form of snow and spring rains. Yearly precipitation from 1980 through 1983 was substantially above average. Soils are moderately deep, well drained, medium textured and are derived from volcanic materials. The study area is a mountain big sagebrush/Idaho fescue (Festuca idahoensis) habitat type. In addition to Idaho fescue, common grasses include Nevada bluegrass (Poa nevadensis), Sandberg's bluegrass (P. sandbergii), bluebunch wheatgrass (Agropyron spicatum), basin wildrye (Elymus cinereus), squirreltail (Sitanion hystrix), and western needlegrass (Stipa occidentalis).

Forbs sampled or known to occur in the area are: false dandelion (Agoseris spp.), tailcup lupine (Lupinus caudatus), yarrow (Achillea millefolium), goatsbeard (Tragopogon dubius), microseris (Microseris sp.), willow-weed (Epilobium paniculatum), smartweed (Polygonum douglasii), blue-eyed Mary (Collinsia parviflora), long-leaf phlox (Phlox longifolia), Hood's phlox (Phlox hoodii), onion (Allium sp.), common dandelion (Taraxacum officinale), Wolly eriophyllum (Eriophylum lanatum), clover (Trifolium sp.), tansy mustard (Descurainia pinnata), pigweed (Chenopodium fremontii), stickweed (Lappula redowski), Brewer's navarettia (Navarretia breweri), and fiddleneck (Amsinckia retrorsa).

Mountain sagebrush is the dominant shrub. Other shrubs include green rabbitbrush

(Chrysothamnus vicidiflorus), rubber rabbitbrush (C. nauseosus), horsebrush (Tetradymia canescens), and bitterbrush (Purshia tridentata).

About 400 elk inhabit the Bennett Hills. This population is a result of a successful relocation in the late 1950's. The Johnson Hill elk herd, or desert elk herd, as they are sometimes called, occupy an unusual year-long habitat for elk viz. sagebrush range. The elk and mule deer use the study area from about April through November. There are several sage grouse leks within a 3.2 km radius of our study sites. Grouse use includes nesting and brood rearing.

Cattle and sheep graze the area in portions of the Macon Flat and North Shoshone allotments. Both allotments use rest-rotation grazing systems and pastures receive complete livestock rest once every 3 years. Total livestock use has been substantially down in the last 5 years.

The study sites were treated in 1980, 1982, or 1983.

On April 27, 1980, 0.8 ha was burned in the mountain big sagebrush zone.

On May 2, 1980, approximately 400 ha of mountain big sagebrush were aerially sprayed with 2,4-D at the rate of 1.8 Kg acid equivalent chemical per hectare. Water and a wetting agent were used as the carrier. The spray was timed to coincide with Poa sp. being in the "boot" stage of phenology.

In the fall (approximately October 16 to November 4) of 1980, 1982 and 1983, about 800 hectares were also burned according to prescription. The prescribed burning index, a standardized measure of wind speed, temperature, relative humidity, and other variables, was between 12 and 26. We usually chose areas at least 0.8 km from water or often on moderate slopes to avoid livestock concentration areas. We also selected areas where native perennial grasses and forbs were well represented in the understory. The resulting burns were normally between 2 and 16 hectares, irregularly shaped, and with abundant edge.

In 1984 we established nested frequency plots in each of the following areas: 1980 spring burn, 1980 spray, 1980 fall burn, 1982 fall burn and 1983 fall burn. Control plots were established on untreated areas adjacent to each of the treated plots. Care was taken to ensure that soils, slope, and vegetation type of the control areas were nearly identical to the treated areas. In most cases, control plots were within 100 meters of the treated areas.

The nested frequency technique involves identification of plants rooted in each of four nested plots measuring 5 cm X 5 cm, 25 cm X 25 cm, 25 cm X 50 cm, and 50 cm X 50 cm all within one frame. (US BLM, 1984). Eighty frames (320 separate plots) were read in each area. Frequencies of occurrence were analyzed in 2 X 2 contingency tables with a significance level of $P < 0.05$ (Zar, 1974). The null hypothesis for each test was that the frequency of occurrence of a given species was the same in the control plot and treatment plot. We grouped data in 3 categories to rate relative amounts of change (+ or -): < 50 percent change from control, 50-100 percent change from control, and > 100 percent change from control.

To compare forage production with results from the nested frequency technique, we clipped and weighed vegetation in four 1.16 m2 plots in the 1982 fall burn and in the control. Clippings were then air dried and reweighed.

Results and Discussion

Burn

Most forbs responded favorably to burning (Table 1), although the degree of increase varied by species, season, and age of burn. Lupine, a key summer forb for elk, was relatively most abundant four years after a fall burn. Phlox was less frequent both 2 years and 4 years following a burn but had slightly increased 4 years after a spring burn. Several key forage plants for sage grouse were not abundant enough to yield significant data at our level of sampling. We observed, however, that common dandelion and yarrow returned to the spring burn area within 2 weeks following treatment. Within a month, clover was also common.

Frequencies of western needlegrass and Idaho fescue were not significantly different on the burn than the control plots on both 2 and 4 year old burns. Frequency of both squirreltail and bluegrasses decreased on both 2 and 4 year old burns. Based on production estimates, however, western needlegrass production was 5 times that of the control 2 years following a fall burn.

Spray

Most forbs either showed a decrease or no change in frequency 4 years following a spray when compared to the control. We observed that common dandelion, false dandelion, and clover were immediately killed after treatment with 2,4-D and were not again seen until the third season following treatment. Squirreltail and cheatgrass greatly increased.

In southeastern Idaho in a mountain big sagebrush type, Harniss and Murray (1973) found total forbs to have increased after 3 years following an August burn.

Harniss and Murray (1973) and Wright and Bailey (1982) indicate needle and threadgrass (Stipa comata) and Stipa spp. in general are adversely affected by fire.

Table 1: Frequency of selected plants in burned and sprayed areas (evaluated in July 1984) when compared to untreated areas

	TREATMENT			
Species	1982 Fall Burn	1980 Fall Burn	1980 Spring Burn	1980 2,4-D Spray
Lupinus caudatum	+	+++	+	-
Epilobium paniculatum	+++	++		0
Agoseris sp.	+++	0	0	-
Polygonum douglassi	0	++	0	-
Phlox spp.	--	---	+	-
Descuriana pinnata		+++	0	0
Lappula redowski		+++		+
Navarretia breweri		++		
Amsinckia retrorsa		0		
Chenopodium fremontii	0	0	0	+++
Stipa occidentalis	0	0		
Festuca idahoensis	0	0		-
Sitanion hystrix	-	--	++	+++
Poa spp.	-	-	+	0
Agropyron spicatum		0	0	-
Bromus spp.		--		+++
Artemisia tridentata spp. *vaseyana*	---	-	-	0

(0) no significant difference
(-+) 50% change from control
(--) 50-100% change from control
(++) 50-100% change from control
(---) > 100% change from control
(+++) > 100% change from control

All differences are at $P < 0.05$ level.
Voids indicate species was not present in treatment area or control.

Our experience indicates that Stipa occidentalis is benefitted under the conditions of a late October burn. Western needlegrass may have a lower percentage of leafy material than other needlegrasses adversely affected by fire and consequently may be less susceptible to fire damage. The late October burn and abnormally high rainfall may also be factors.

Although our frequency data showed no significant difference between Idaho fescue occurrence in the 2 year old burn and the control, our limited clipping data did show a significant decrease in production in Idaho fescue in the burn plot. Others have documented the susceptibility of Idaho fescue to fire (Wright and Bailey, 1982).

Wildlife Benefits

Rocky mountain elk diets vary considerably geographically and seasonally, but most researchers agree that grass is their most important spring forage. As the grasses dry in summer, elk shift to forbs and shrubs (Nelson and Leege, 1982).

If sage grouse cover requirements are met (Braun et al., 1977) and key winter ranges and nesting areas are not involved, prescribed burns conducted in small random patterns can have a positive impact where forb production is enhanced for brood use.

Sage grouse feed heavily on forbs in the spring and early summer (Autenrieth 1981). In our study area, key upland forbs start drying out by the end of June and by mid-July grouse probably have shifted to wetland areas for succulent forbs.

If one considers just the forage needs of elk and sage grouse, increases in forbs could benefit both species. Burning in our study increased both forbs and several grasses as compared to spraying which generally favored just grasses. The Western States Sage Grouse Committee (1982) attributed high sage grouse brood success to a longer availability of succulent forbs. Blaisdell and others (1982) observed vegetation in burn areas remained green longer than in similar untreated areas.

Recommendations and Conclusions

We feel that burning is clearly the treatment of choice compared to spraying in our study area for grouse and elk habitat improvement. Grasses and forbs are enhanced while brush competition is reduced. The same probably holds true for livestock also, especially when considering the costs of both treatments.

Baisdell and others (1982) provided excellent guidelines for using fire in sagebrush/grass ecosystems and we refer the prospective burner to these. Based on our experience we would add and emphasize several of them.

1. Fall burning is most practical. In the Shoshone District the last 2 weeks in October appear on the average, to be the best time for burning. We have had limited success with spring burns. In many years greenup immediately follows snowmelt.

2. Burns in areas 0.8 km or more from livestock water or on slopes less than 30% have a less chance of being "camped on" by livestock following the rest period.

3. The burn areas should be small enough to provide suitable nearby cover but sufficiently large enough to prevent rapid establishment of sagebrush from seed of adjoining unburned plants. Small burns (1-2 ha) although ideal for sage grouse, tend to be disproportionately trampled by livestock and will quickly become reestablished with mountain big sagebrush. Numerous small burns distributed throughout the area and delayed grazing would reduce livestock impacts by spreading use and allowing sage grouse broods to use spring forbs when their importance is greatest.

4. We cannot overstate the importance of selecting areas for burning with a good understory of desirable plants. Limited positive results can be expected in areas with less than 30 cm annual rainfall, especially where cheatgrass (Bromus tectorum) and/or rabbitbrush may become established.

5. The quick return and growth of forbs following burning demonstrates the key variable between 2,4-D spray and prescribed burn treatments.

Acknowledgement

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Studying Rangeland Fire Effects: A Case Study in Nevada

Greg A. Zschaechner

Abstract

Northern desert shrub plants were monitored on four prescribed burns in Nevada. The burns were conducted in August and October, 1980, and follow-up plant response information gathered for four years postburn. Actual plant responses are compared with expectations based on reviews of the fire effects literature for each species examined. Similarities and differences between expected and actual plant responses are discussed with reference to the fire behavior and other factors related to the burns. A conceptual flow chart of the process used in predicting fire behavior and effects is presented. The fires are displayed on the Fire Behavior Fire Characteristics Chart.

Introduction

In 1980, the Nevada State Office, Branch of Protection of the Bureau of Land Management initiated a study to calibrate fire behavior prediction models in the sagebrush/grass fuel type and at the same time to observe fire effects in this plant community. Most often past studies of fire effects have focused on after-the-fact situations, where fire behavior information was not available or not discussed. In a few instances fire behavior was discussed in subjective terms such as "fast", "hot", "cool" or "running". This hinders meaningful correlation of actual fire behavior to fire effects. These general terms make it nearly impossible to compare the results of different studies (Rothermel and Deeming 1980).

This paper attempts to bring current information from fire effects literature together with anaylsis of preburn and postburn site conditions and actual fire behavior observations. Plants observed in this study include: big sagebrush (Artemisia tridentata), antelope bitterbrush (Purshia tridentata), green rabbitbrush (Chrysothamnus vicidiflorus), rubber rabbitbrush (Chrysothamnus nauseosus), mountain snowberry (Symphoricorpos oreophilus), Utah serviceberry (Amelanchier utahensis), bluegrass (Poa spp.), needlegrass (Stipa spp.), bluebunch wheatgrass (Agropyron spicatum), bottlebrush squirreltail (Sitanion hystrix), basin wildrye (Elymus cinereus), tailcup lupine (Lupinus caudatus), arrowleaf balsamroot (Balsamorhiza sagittata), stickseed (Hackelia spp.), fiddleneck (Amsinckia spp.) and horsemint giant hyssop (Agastache urticifolia).

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Guiding Concepts

Rothermel and Deeming (1980) proposed methods of describing fire behavior using terms that are measurable and predictable in uniform and porous fuelbeds, such as grass or shrubs. These methods were selected for this study since they applied to Nevada fuels, the data needs and collection methods were clearly identified, and the product would be presented in a standard, quantitative form.

Wildland fire managers are concerned with the accurate prediction of fire effects on the land resource. By including analysis of actual fire behavior with observation and documentation of fire effects on vegetation, fuels, and other resources, these effects can be predicted with reasonable accuracy, taking into account burning conditions and other ecological factors. By correlating fire behavior with fire effects, we can extend our prediction capability by applying Rothermel's (1972) predictive model of fire behavior. Given inputs of fuel type, cloud cover, air temperature, relative humidity, dead fuel moisture, live fuel moisture, slope and windspeed, the model generates descriptors such as flame length, heat per unit area, and fireline intensity. These quantitative descriptors can then be correlated with fire effects (Range and others 1982).

For example, a resource manager learns, after observing several wildfires and/or prescribed burns, that a certain flame length and rate of spread (fire behavior) results in complete removal of big sagebrush cover (fire effect). With a fuel model, the manager can find the proper combination of windspeed and live and dead fuel moistures (burning conditions) that generate the kind of fire behavior that, in turn, produces the desired effect.

Figure 1 combines these three concepts - burning conditions, fire behavior, and fire effects. The Burning Condition subsystem describes the site in terms of fixed and variable elements that influence the way a fire burns. A fuel model is first selected from one of thirteen Northern Forest Fire Laboratory (NFFL) fuel models (Anderson 1982) or a custom model is developed through BEHAVE (Burgan and Rothermel 1984). The fuel model describes the fuel arrangement, loading, chemical make-up, etc. Variables that are site specific are then entered into the model as input.

The Fire Behavior subsystem is concerned with the prediction of fire behavior and the observation and description of actual fires. Fuel model predictions of fire behavior are tested for accuracy against actual observations, and adjustments are made, accordingly.

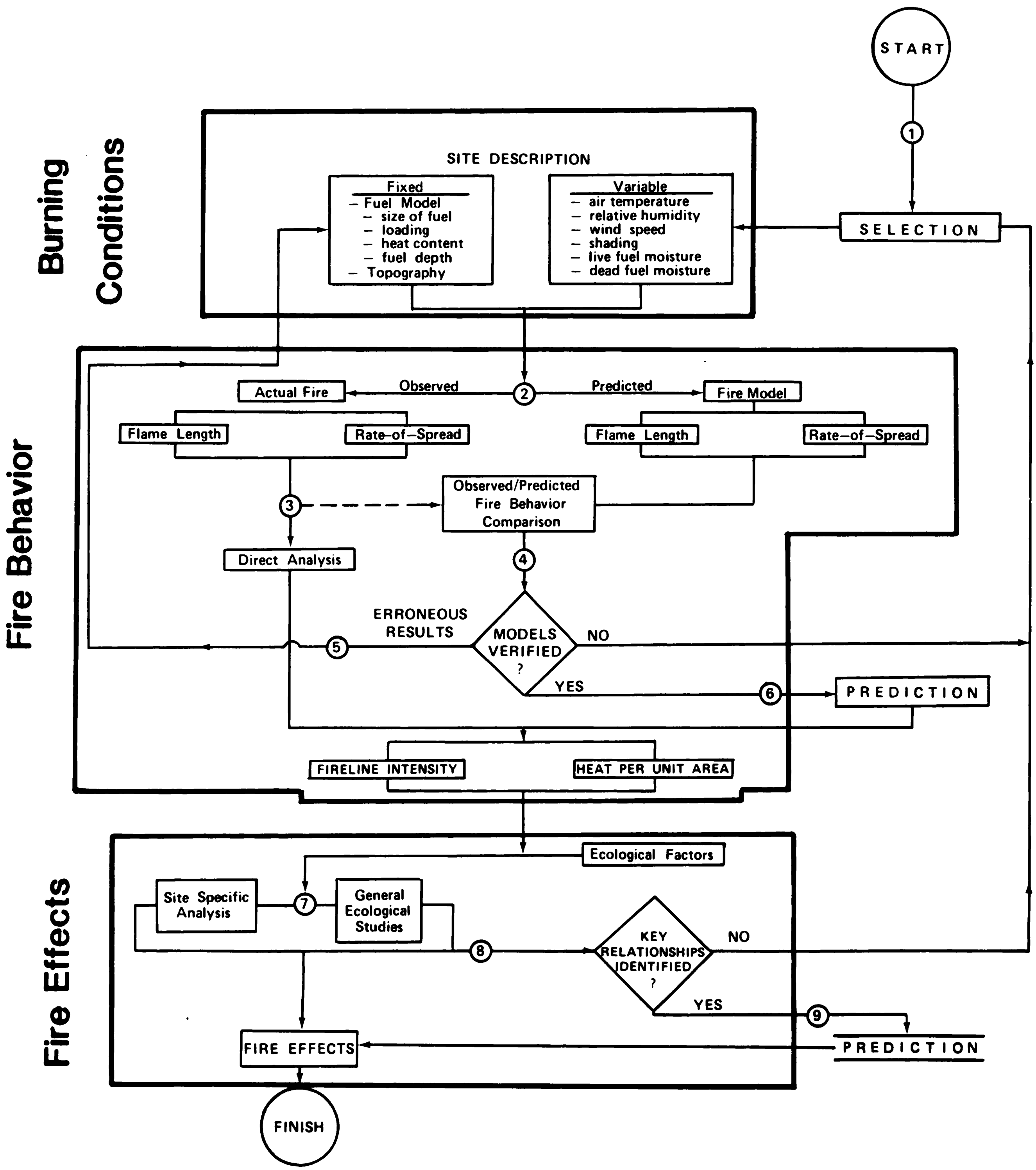

Figure 1. Fire behavior and fire effects prediction flow chart.

In the Fire Effects subsystem, fire behavior descriptors (fireline intensity and heat per unit area) are combined with ecological factors to analyze fire effects. The goal of predicting fire effects from a given set of burning conditions is achieved by first predicting the potential fire behavior, then by assessing the impact of that fire behavior plus other ecological factors on the final outcome. Key steps in this process are identified by numbers 1 through 9 in Figure 1.

Step 1 involves selection of a fuel model and the conditions under which a prescribed fire will be carried out. In a wildfire situation this could be the conditions under which a fire is observed to be burning. In a planned fire, this is referred to as a burning prescription -- a set of variables whose values must be within a specified range before ignition can occur. After several fires in a particular fuel type a manager will develop confidence. With increased confidence and experience, the burn prescriptions can be refined to achieve more specific objectives. For more information on this topic see Fisher 1978, Martin and others 1978, Green 1981.

In Step 2 a wildfire is monitored or a prescribed fire is ignited (Observed). Inputs are made for the fuel model (Rothermel 1972), which then returns fire behavior predictions (Predicted). The observed fire behavior can then be converted into quantitative fire behavior descriptors of flame length and rate of spread, either for direct analysis of that particular fire, or as Step 3 directs, these descriptors can be compared with the ones predicted by the fuel model.

At Step 4 a decision must be made. Are the predicted and observed parameters being met to your standards? If not, the cycle of selecting the burning conditions, making fuel model inputs, igniting observation fires, and comparing the observed and predicted values is repeated until the answer to this question in Step 4 is YES. Prediction of fire behavior, given specific burning conditions, is then possible -- Step 6 - FIRE BEHAVIOR PREDICTION. If the results found in Step 4 are erroneous, several alternatives exist. A new fuel model may have to be selected, correction coefficients may be assigned to adjust the fire behavior outputs, or internal adjustments may be made in surface-to-volume ratios, fuel loading, etc. in the BEHAVE-CUSTOM program.

At this point, Step 6, fire behavior can be described in quantitative terms for the purpose of explaining and comparing fire effects. Two aspects of fire behavior can be related to fire effects: fireline intensity and heat per unit area (Rothermel and Deeming 1980). These elements will be discussed in more detail in the next section.

In Step 7, fire behavior information is combined with ecological factors that may account for the observed fire effects. As in Step 3, there is an opportunity to bypass analysis for prediction purposes, and instead concentrate on site specific information. This route still offers the opportunity for input of the specifics for general ecological studies, which can be geared to address Step 8. For example, the objective of the prescribed burn is fuel hazard reduction. Few key relationships exist and little ecological information is necessary to evaluate the effectiveness of a particular fire behavior in reducing fuel loading. However, should the manager also require a forecast of postburn vegetal recovery, the same fire behavior and fuel consumption data can be applied to analysis of soil nutrient studies, seedbed preparation, or other fire related subjects.

The key relationships referred to in Step 8 are those that exist between fire behavior, ecological factors, and the resulting fire effects. This cycle continues until plant response is predicted, Step 9. It is critical to point out that many of the interactions occurring in the plant community are subject to the element of change, for example rainfall. In these situations there may be a need for several fire effects outlines. A set of alternative outcomes addressing combinations of possible events would be more realistic. For example, one could predict that following a burn for sagebrush removal forbs would increase a certain percentage over the burn if the following year was wetter than normal; but this increase would be less, or absent, if a drought year followed.

In summation, predicting fire effects is not a simple task. It requires one to analyze the weather and fuels, to make fire behavior predictions and actual fire observations. It combines this information with ever changing ecological factors. The accuracy of any fire effects forecast depends on the quality and quantity of information documented for each natural community. It is important that every effort is made to apply consistent and objective techniques in the study of fire and its effects.

Procedures, Interpretation Tools, and Area Descriptions

Measurement of Plant Response

On each burn site several species of shrubs, grasses and forbs were marked and photographed. Iron stakes made from reinforcement bar were used to distinguish the plants being monitored. These stakes were cut at 30-cm in length and stainless steel number tags attached. The author now recommends the use of two per plant with the plant centered between the stakes. Shrub centering serves several purposes: plant positions are less likely to be lost due to an offset stem from the crown; rodents may knockdown one stake leaving one to relocate the plant; and plants are easier to align in photographs using the stakes as guides. Figure 2 illustrates this process.

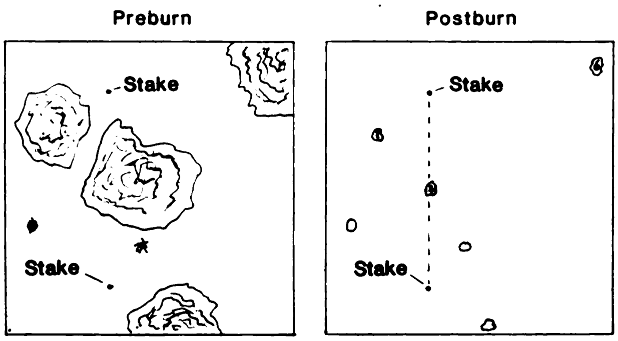

Figure 2. Preburn and postburn views of staked plant. Preburn illustrates the location of stakes equidistant from the plant base. The plant can then be easily located postburn by extending a line between the stakes. With only one stake marking a plant, location of the exact plant being monitored may be difficult.

Preburn measurements made for shrubs include phenological stage, crown diameter (maximum crown diameter and diameter perpendicular to the maximum crown diameter), shrub height (maximum height and height at maximum crown diameter), basal diameter, a visual estimate of percent of dead crown volume, litter depth in quadrants surrounding shrub base, and surrounding shrub rating in three classes: shrubs at distances greater than one meter, shrubs less than one meter, and shrubs actually in contact with the plant being monitored.

Perennial grasses were measured for total height, basal area, litter depth and phenological stage. A small number of individual species were sampled as a compromise, given the time available for sampling.

Percent cover and species composition were sampled with line intercept transects (Canfield 1941). This method was modified from an overlapping to a non-overlapping inventory since it was primarily used for biomass determination. Litter weights were obtained by removing the material within a .10-m^2 frame, oven-drying, and weighing.

Immediately after the burn, the plants were again photographed, and the degree of shrub consumption was recorded as defoliation, consumption of stemwood in 0 to 0.6-cm, 0.6 to 2.5-cm, and 2.5 to 7.6-cm classes, complete consumption, or unburned. Bunchgrasses were noted as defoliated or completely consumed. Ash depths at the bases of the plants were measured and color noted as black, grey, or white.

Following the 1980 burns, the sites were resampled at eleven, twenty-two, thirty-six and fourty-eight months. At each period line transects were measured to determine species composition change and the marked plants were photographed and recorded as dead, surviving, resprouting, or seedlings present. Plant measurements were also taken.

Measurements of Fire Behavior

Rate of Spread and Flame Length - Observations of fire rate of spread and flame length were used to determine two different forms of fire intensity: fireline intensity and heat per unit area (Rothermel and Deeming 1980).

Fireline intensity (I) is calculated from flame lengths (F_L) in a headfire by the equation:

$$I = 5.67\ F_L\ 2.17$$

Fireline Intensity is expressed as a rate of heat energy released by each foot along the edge of an advancing fire front (Btu/ft/sec) (Byram 1959). Flame length is the distance, in feet, from the flame tip to its base, midway in the flaming zone of the fuelbed (Figure 3).

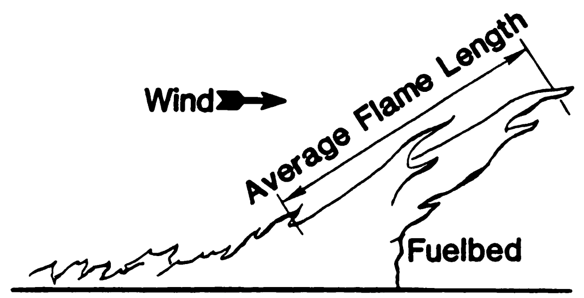

Figure 3. Flame dimensions shown for a wind-driven fire.

Heat per unit area (H/A) relates the total amount of energy released in the flaming zone by burning a unit area of a given fuelbed. It is obtained by dividing the fireline intensity by the fire's rate of forward spread (R).

$$H/A\ \text{Btu/ft}^2 = \frac{(60)\ (I)\ \text{Btu/ft/sec}}{(R)\ \text{ft/min}}$$

Flame lengths were estimated by two methods, direct observation and infrared photographs taken during the burn using 6 foot reference placards and a stopwatch timer (Britton and others 1977). Fire rate of spread was measured by timing the fire's advance over a known distance. Markers were tossed at the beginning of a fire run, timed and marked again with a marker at the end of the run. The line transects used for vegetation sampling also served as baselines for registering fire behavior observations with the locations of the tagged plants.

Fire Temperature - Maximum fire temperatures were estimated using heat-sensitive lacquers (Tempilaq) painted on metal strips and placed at the following levels: 1, 2, and 3-m above ground, at ground surface, and at 1 and 2-cm below the ground.

Measurement of Burning Conditions

Burning conditions are defined as those factors which influence fire behavior, the fuels and topography of the site, weather, and fuel moistures at the time of the burn.

Fuel Loading - Fuel loadings were estimated by translating percent cover of shrubs into tons per acre using averages of plant sizes and weights (Range and others 1980).

On four randomly placed 10-m^2 circular plots, shrubs were cut at ground level, separated, and weighed by size classes: foliage; live stems 0 to 0.6-cm, 0.6 to 2.5-cm, and greater than 2.5-cm; and dead stems 0 to 0.6-cm, 0.6 to 2.5-cm, and greater than 2.5-cm. The entire crown of a shrub was sampled if the base of the plant fell within the plot. Live and dead fuel moistures were determined with an automatic weight loss type moisture analyzer (Computrac moisture analyzer), and the values obtained were used to correct field weights of shrubs to oven-dry weights. Fuelbed height was determined by taking 0.8 of the average large plant height (Brown and others 1982).

Weather - The air temperature, wind direction, windspeed, and relative humidity were measured with hand-held devices provided in the belt weather kits. A continuous recording type unit (Climatronics) was also used in addition to the hand-held devices.

Interpretation Tools - Until recently, fireline intensity has been poorly understood, with no real descriptors (Albini 1976). Technically, "intensity" relates to some measure of energy transmission. Firefighters and fire researchers have used "intensity" to describe many aspects of fire behavior and fire effects such as peak flame temperature, maximum soil temperature, etc. Fireline intensity (Btu/ft/sec) has a close correlation with flame length, it represents what most people seem to visualize when they speak loosely of fire intensity. In Table 1, flame length and fireline intensity are interpretated in suppression terms (Andrews and Rothermel 1982).

Heat per unit area (Btu/ft^2) is a measure of the amount of heat being released by a square foot of fuel while the fire is passing that area. This is a rough measure of the impact that a fire has on a site one square foot in size. Since most wildland fuels produce about the same amount of heat when burned, the total heat released by burning relates well to the amount of fuel consumed.

Heat per unit area is a good indicator of the amount of heat that is transferred to the soil surface during a fire. Fires that have a low rate of spread have a longer residence time over a unit area, thus causing more fuel to be consumed and more heat to be released there. A fast moving fire would have a short residence time and would have less heat released. This information would be useful in determining fire's impact on soils, plant root systems, etc.

All of these fire behavior descriptors (flame length, rate of spread, fireline intensity and heat per unit area) can be combined on to a chart. Andrews and Rothermel (1982) created the Fire Behavior Fire Characteristics Chart (Figure 4). The chart displays these four fire characteristics graphed as a single point.

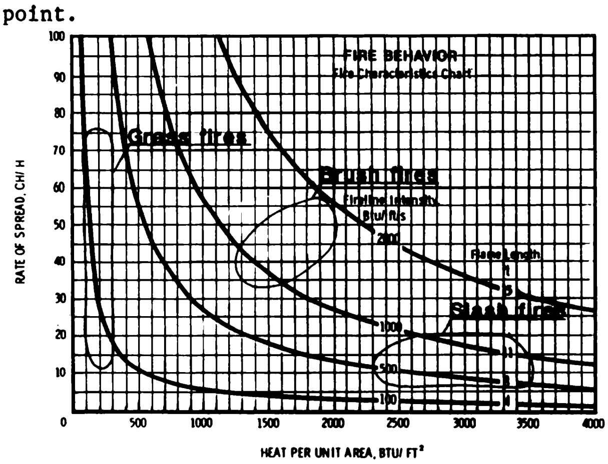

Figure 4. Fire behavior fire characteristics chart showing characteristics of grass, brush and slash fires.

Table 1. - Fire suppression interpretations of flame length and fireline intensity.

Flame length	Fireline intensity	Interpretation
Feet	Btu/ft/s	
less than 4	less than 100	Fire can generally be attacked at the head or flanks by persons using handtools. Handline should hold the fire.
4-8	100-500	Fires are too intense for direct attack at the head by persons using handtools. Handline cannot be relied on to hold fire. Equipment such as plows, dozers, pumpers, and retardant aircraft can be effective.
8-11	500-1,000	Fires may present serious control problems - torching out, crowning, and spotting. Control efforts at the fire head will probably be ineffective.
more than 11	more than 1,000	Crowning, spotting, and major fire runs are probable. Control efforts at head of fire are ineffective.

The position of a plotted point on the graph shows the severity of a fire in two ways. If heat per unit area is taken as a measure of severity as in fire impact on soil or in fuel hazard reduction, then fires plotted further to the right are more severe. If rate of spread is the critical factor, as in fire control situations, points highest on the graph are of great concern. For example, a fast spreading grass fire of low intensity would plot near the vertical axis, whereas a high intensity, slow moving slash fire would lie close to the horizontal axis. The further a point lies to the upper right of the graph's origin, the more severe the fire.

Fires are represented by single points on the chart, but this is only an estimate of fire behavior; therefore, a circle would more truly represent the uncertainties of the calculation. It must be remembered that the key component in the chain of events (burning conditions - fire behavior - fire effects) leading to prediction of fire effects is the predictive model of fire behavior (Rothermel 1972). For the model to deal with the complex phenomenon of fire spread, certain simplifying assumptions are made. The most significant of these is that the fire spread is through continuous, uniform fuels, such as tall grass or a layer of pine needles. This ideal fuelbed rarely occurs over extensive areas in the wildlands. Patches of shrubs may occur within the grassland, or dense piles of downed wood may be found about the forest floor, causing the fire to flare up as it moves from the uniform fuels into these areas. Thus, the more discontinuous or patchy the fuels are, the less accurate the output from the fuel model, and the larger the plotted circle should be on the fire characteristics chart.

Observers may wish to monitor fire behavior for fuel model verification or for determining fire effects. Fuel model verification requires an average observation of the flamming front under known fuel and weather conditions. The level of monitoring of fire behavior for fire effects will depend on the degree of sensitivity desired. Fire behavior can be monitored on a plant by plant basis or on a more general basis over an entire site. This study used both techniques in an attempt to tie the fire behavior immediately surrounding a plant to its postburn response and to verify the fuel model.

Description of Sites

The Jackpot burn site is located on the Elko District at T.46N, R.63E, Section 10 (Figure 5). This is approximately 8 miles southwest of the town of Jackpot, Nevada. Four small drainages are contained on the 150 acre site. Aspects are generally to the north, slopes are approximately 7 percent and the elevation is 2,010-m (6,600 ft).

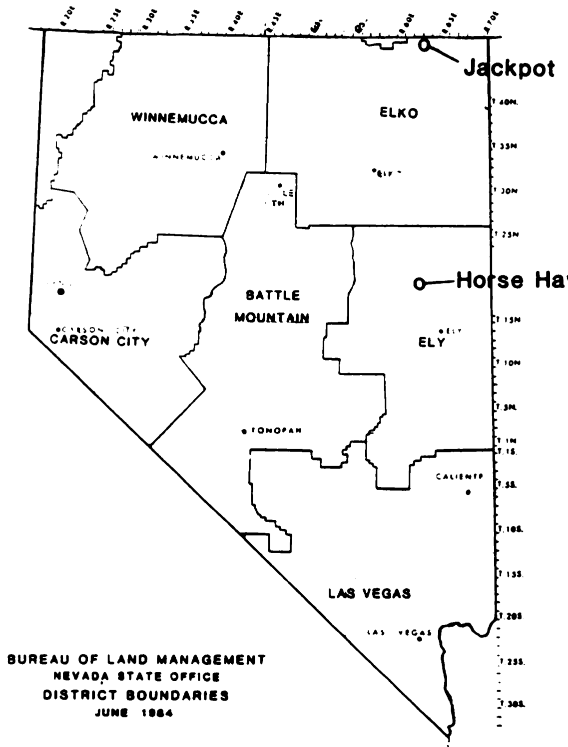

Figure 5. Study site locations.

The climate of the area may be seen in Figure 6. The graph represents the mean precipitation and fluctuations occurring during the years of 1980 through July 1984.

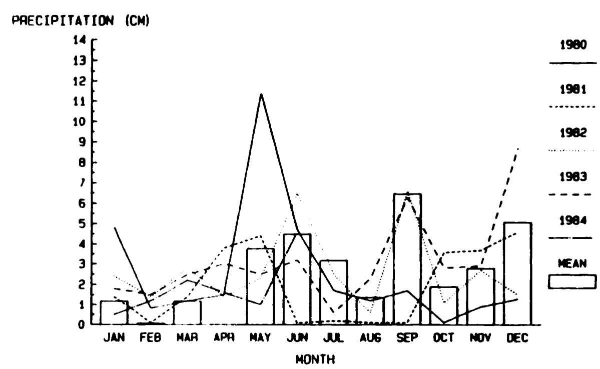

Figure 6. Precipitation for Jackpot site, elevation 2,010 meters. Weather information taken at Contact, Nevada, elevation 1,635 meters.

Soils are estimated to be members of the five loamy and clayey families of Torriorthentic Haplexerolls in the drainage bottoms, Typic Argizerolls on the lower sideslyses, and Xerollic Haplargids with rock outcrops on the ridges.

Preburn vegetation consisted of big sagebrush, low sagebrush, mountain snowberry, green rabbitbrush, rubber rabbitbrush, antelope bitterbrush, Utah serviceberry, bluegrass, needlegrasses, bluebunch wheatgrass, basin wildrye, and a few forbs. The stand age was estimated by ring counting to be approximately 45 years.

The Horse Haven burn site is located on the Ely District in eastern Nevada at T.19N, R.62E, Sections 27 and 28. This is approximately 16 miles northwest of the town of Ely, Nevada (Figure 5). The site is 40 acres in size and has a southwest aspect. The slope is 12 percent at an elevation of 2,285-m (7,500 ft). Figure 7, shows the general precipitation for the area.

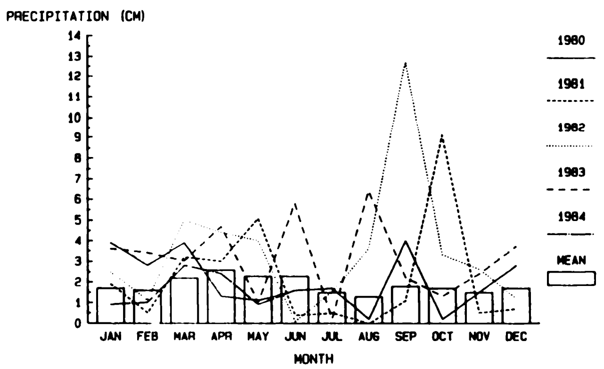

Figure 7. Precipitation for Horse Haven site, elevation 2,285 meters. Weather information taken at Yelland Field, Ely, Nevada, elevation 1,905 meters.

Soils are derived from quartzite and volcanic rocks and are classified into the Simme series which is a member of the loamy-skeletal mixed frigid family of Aridic Argixerolls, with associated Mascump series (loamy-skeletal mixed frigid Aridic Lithic Argixerolls) and rock outcrops. The depth to bedrock is about 66-cm.

Vegetation at the Horse Haven site is made up of the same species as found at the Jackpot site. The stand age was estimated to be 35 years.

Results and Discussion

General Fire Behavior

The Fire Behavior Fire Characteristics Chart (Figure 8) illustrates the observations of flame length and rate of spread for all the burns. Each run of the fire on which rate of spread was measured was associated with a particular flame length. Refer to Table 2 to find the burning conditions responsible for the fires behavior.

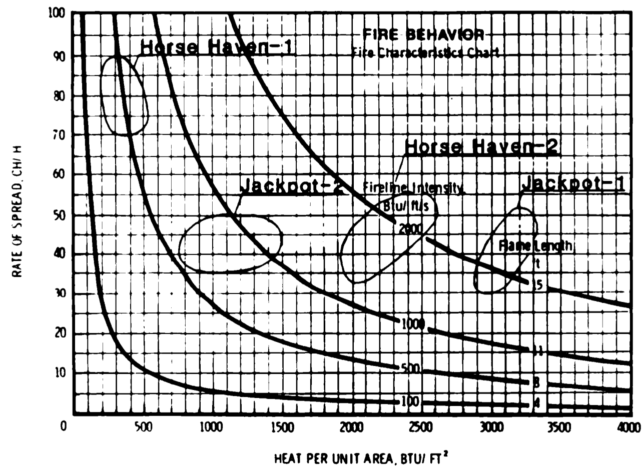

Figure 8. Fire characteristics chart for study sites.

Jackpot Burns 1 and 2, and Horse Haven Burn 2 were marked by erratic fire behavior. Although fuels were uniform over a distance of 10-m, windspeed and direction changed rapidly and the ideal fire (steady state) never occurred. At Jackpot, burning within the narrow drainages, with a heavy load of fire fuels in the shrub understory, accounts for these results. The angle of the slope relative to the wind at Horse Haven-2 and sequences of gusts and lulls were factors responsible for the lack of steady state fire on that site.

Horse Haven-1 permitted the best data collection of all burns. On this burn the fire intensities were monitored more closely and can be associated with zones within the burn.

Table 2. Burning Conditions.

Site	Burn Date	Time	Air Temp (°F)	Relative Humidity (%)	Wind Speed (mph)	Fuel Moisture (%) Live	Fuel Moisture (%) Dead	Fuel Depth (cm)	Fuel Loading (lb/ac)	Soil Mois. (%)
Jackpot-1	08/27/80	1100	74	24	5	92	4	61	3.48	7
Jackpot-2	10/06/80	1130	70	27	5	77	9	52	*	10
Horse Haven-1	08/29/80	1400	89	14	8	92	4	70	3.03	7
Horse Haven-2	10/08/80	1300	74	16	3	77	5	88	3.50	*

* Information not recorded.

Vegetational Change

Figures 9, 10, 11, and 12 illustrate vegetation transect data for Jackpot-1, Jackpot-2, Horse Haven-1, and Horse Haven-2 burns, respectively. The years of 1980 through 1984 are shown. It should be noted that a non-overlapping transect was used primarily for determination of fuel loading. For this reason, postburn recovery for grasses and forbs may be slightly exaggerated.

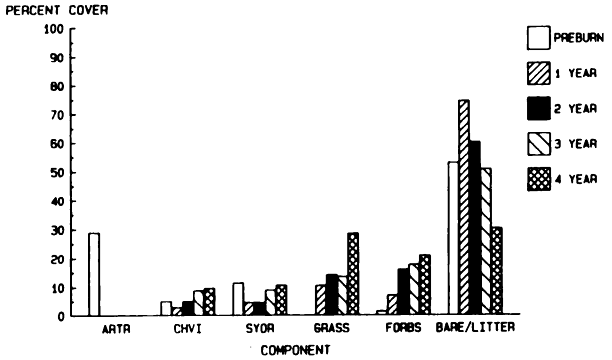

Figure 9. Jackpot-1 vegetation cover for preburn and four years postburn.

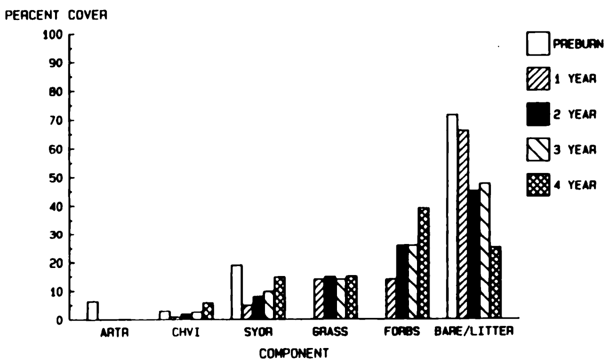

Figure 10. Jackpot-2 vegetation cover for preburn and four years postburn.

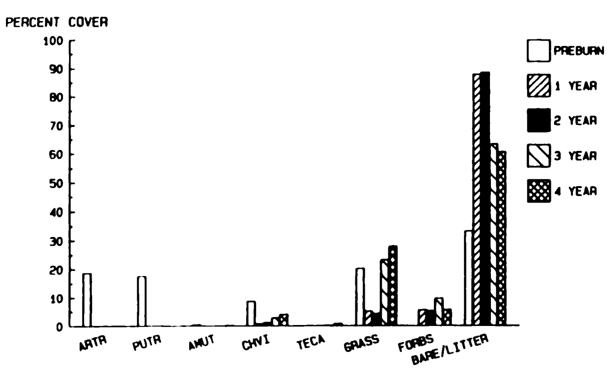

Figure 11. Horse Haven-1 vegetation cover for preburn and four years postburn.

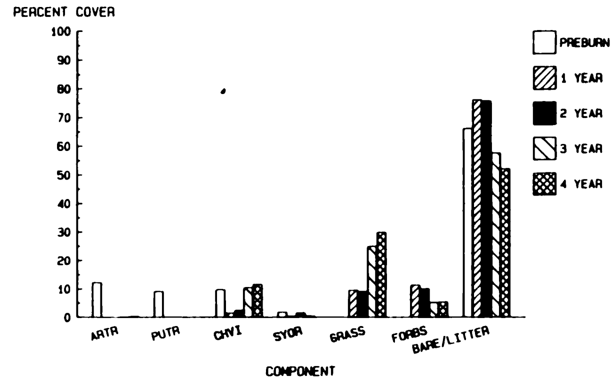

Figure 12. Horse Haven-2 vegetation cover for preburn and four years postburn.

On all the burns, the late summer fires consumed all but a trace of big sagebrush and litter. These two components showed zero cover after four seasons postfire. Inversely, bare ground showed high increases. The sprouting shrub, green rabbitbrush, showed good recovery on the Jackpot-1 and 2 sites; but poor recovery on Horse Haven-1 and 2 sites. Seventy-five percent of the plants died on the Horse Haven-1 burn.

Antelope bitterbrush suffered 100 percent mortality of the plants tagged. Several individuals were observed sprouting two years postburn. By the fourth year a number of seedling were observed to be growing vigorously.

Mountain snowberry, a known sprouter, experienced good recovery. On Horse Haven-2 it reached preburn levels by the second season postburn and then was browsed back. On Jackpot 1 and 2 snowberry has reached 93 and 78 percent, respectively, of the original cover.

By the fourth year all four study sites showed grasses as having a tremendous increase. On Jackpot-1, the bluebunch wheatgrass had the most cover for grasses, with Idaho fescue and Sandberg bluegrass contributing lesser amounts. Bottlebrush squirreltail, Columbia needlegrass, oniongrass, basin wildrye, and cheatgrass were also present. Columbia needlegrass was the most common grass on Jackpot-2 at 8.4 percent. Other grasses that increased in value were Idaho fescue, Sandberg bluegrass, and oniongrass. On Horse Haven-1, cheatgrass has shown a large increase to 24.0 percent with bottlebrush squirreltail at 5.1 percent. On Horse Haven-2, Thurber's needlegrass (<u>Stipa thurberiana</u>) has increased 8.1 percent with bottlebrush, cheatgrass and Sandberg bluegrass (<u>Poa secunda</u>) at lesser amounts.

The first year, forbs showed increased cover on all burn sites. During the second year forbs only increased on Jackpot-2 and remained equal to year 1 on the other sites.

The third year forbs increased greatest on Horse Haven-1 and 2. By the fourth year there was a decrease at Horse Haven-1 and 2 and an increase at Jackpot-1 and 2. At Jackpot, giant hyssop and lupine were very conspicuous during the four years and at Horse Haven Watson penstemon (Penstemon watsonii) was observed to be very abundant the first year with a dramatic decrease the second year. Wayside gromwell (Tithospermum ruderale) and milkvetch (Astrogalus spp.) were observed the remaining three years.

Plant Responses - Expectations and Results

Expectations - For each species studied, a brief literature reivew on its fire effects is presented. Based on this information, expectations regarding the plants' responses to the burns are given.

Burn Results - The outcome of the burns includes the following data:

1. description of individual plants observed (number, dimensions, etc.)
2. observed fire behavior in the plants' vicinity
3. immediate postfire plant condition (consumption class, ash color, etc.); and
4. condition of the plants at the 1981, 1982, 1983 and 1984, sampling date (dead, resprouting, etc.).

Big sagebrush (Artemisia tridentata) (ARTR)

Expectations - Big sagebrush is highly susceptible to fire injury and does not sprout from the stem or root crown following fire (Blaisdell 1953; Pechanec and others 1954; Johnson and Payne 1968). Differences in recovery rates may be related to season of burn as it affects seed production, summer precipitation, and completeness of burn (Wright and Bailey 1982; Kozlowski and Keller 1966). When perennial grasses and weeds are present, big sagebrush seedlings experience increased difficulty in surviving due to the greater vegetative density and less soil moisture available for germination (Pechanec and others 1954). If a good moisture year occurs shortly after burning, big sagebrush reestablishment can be greatly accelerated (Sneva 1978).

Big sagebrush was expected to be temporarily eliminated from the site due to the intense late summer burn and the tiaing of consumption of the pre-flowering buds.

Horse Haven Burns 1 and 2 Results - A total of four big sagebrush and two low sagebrush plants were monitored on the two burns. Their heights ranged from 60 to 130-cm and crown areas varied from .30 to 1.2-m^2. An average of 50 percent of the crown volumes were assumed as dead.

Five plants were burned (flame lengths: 6 to 20 ft.; H/A = 300 to 5,660 Btu/ft^2) and killed, as expected. Two of these resisted complete consumption, with stems greater than 2.5-cm remaining. Apparently, moisture content of new foliage which was nearly 85 percent (both burns) is low enough to permit nearly complete combustion of the big sagebrush. One plant of low sagebrush escaped the fire and was still living the second summer after the burn.

Precipitation was slightly above normal (Figure 7) during the fall after the burn. Nevertheless, no seedlings were observed one year after the fire. Three years after the fire numerous seedlings were observed.

Jackpot Burns 1 and 2 Results - Individual big sagebrush plants (var. vaseyana) were observed, one on the first burn and two on the second. The first plant was 140-cm tall, with a crown area of 2.5-m^2; the other two plants had heights of 140 and 70-cm, with crown areas of 2 and .5-m^2, respectively. Crown volumes contained about 20 percent dead material.

As expected, the plants were killed on both burns. The plant in burn 1 was completely consumed despite the 98 percent moisture content of new foliage. Both plants in burn 2 had stemwood 2.5-cm in diameter and greater remaining. Higher density of surrounding fuel in the burn 1 plant over those of burn 2 may account for these results.

Precipitation records indicate a fall season drier than normal for the Jackpot site (Figure 6). This, combined with the destruction of the big sagebrush flowering buds in the late season burn and the fact that no seedlings were observed in 1981, indicates that reestablishment may be delayed for a few years. In 1984 numerous sagebrush seedlings were observed on both burn sites.

When foliage fuel moistures drop below 120 percent most of the stem is consumed by fire.

Antelope bitterbrush (Purshia tridentata) (PUTR)

Antelope bitterbrush commonly suffers high mortality following fire, especially in the western Great Basin (Billings 1952; Nord 1965; Klebenow and others 1976). Sprouting ability of the columnar growth form, present on the Horse Haven site, is more dependent on fire intensity and soil moisture than that of the decumbent growth form (Monson and Christensen 1975). Hence, the midsummer burn, with low fuel moistures generating high fire intensities, combined with dry soils, was expected to kill most of the antelope bitterbrush plants on the site (Blaisdell 1953; Blaisdell and Mueggler 1956; Pechanec and others 1954).

Natural regeneration was anticipated from rodent-cached seeds (Klebenow and others 1976), and seedlings were expected to survive despite high surface temperatures on the newly bared soil (Ferguson 1972).

Horse Haven Burn 1 Results - Five mature antelope bitterbrush plants were tagged prior to the burns. These ranged from 60 to 120-cm in height and 10 to 40 percent dead crown volume. The plants were in fruiting stage at the time of the burn.

A headfire burned the plants with their foliage moisture at 96 percent and with soil moisture at 6 percent. Flame lengths averaged 6 feet with rates of spread ranging from 25 to 100 feet per minute. These values indicate moderate to high fire severity.

The plants were defoliated, one from radiation alone; however, because of the high foliage moisture content and of the stemwood density, the crowns resisted consumption. Surface conditions at the base of the plants ranged from unburned litter to 2-cm of white ash. No green material was observed on the plants, so all plants along the transects were assumed dead immediately postfire. One untagged plant located in the blacklining operation was observed to be sprouting one year postburn.

By 1984 many plants were observed starting from seed. Several plants were measured at 19-cm in height.

Horse Haven Burn 2 Results - The three antelope bitterbrush plants observed were exposed to over ten times the heat load of the plants on burn 1. Besides killing the plants, the fire consumed stemwood up to 0.6-cm in diameter with fireline intensity reaching 3,770 Btu/ft/sec and heat per unit area values of 5,660 Btu/ft (flame lengths 6 to 20 ft, rate of spread 40 ft/min). No seedlings were observed upon revisiting the site in 1981. Several seedlings were seen in 1984.

Jackpot Burns 1 and 2 Results - No antelope bitterbrush was found on the site.

Green rabbitbrush (Chrysothamnus viscidiflorus) (CHVI)

Expectations - Following fire, green rabbitbrush sprouts from the roots and increases in density through seedling establishment (Young and Evans 1974). Largely dependent on the plant stage at the time of burn, a green rabbitbrush plant may or may not have already deposited seed. This will influence the time required for plant reestablishment. If a fire occurs prior to seed dispersal, plant reestablishment may be slow during the first three years. Beyond the second year seed production will accelerate resulting in greater plant density during the third year (Blaisdell 1953). The literature reviewed did not address the effects of varying fire intensity and fluctuations in annual precipitation on green rabbitbrush survival. Nevertheless, the plants were expected to sprout during the year following the fire, produce seed, and by the fourth year to exhibit rapid growth as an even-aged stand.

Horse Haven Burn 1 Results - Four individuals of green rabbitbrush were tagged before burning. These plants could be considered subshrubs, with heights of 40-cm and crown areas less than 1/4-m^2. Three of the plants were in a budding stage and one was still vegetative. Litter depths averaged 3-cm at the plant bases.

Rates of spread from 25 to 100 ft/min and flames from 6 to 8 feet long were recorded. Three plants were totally consumed by fire with a rate of spread less than 50 ft/min. The plant burned by 100 ft/min fire spread had only its foliage consumed. Although the foliage and twig moisture cbntents were over 100 percent oven-dry weight, the flammable resins in rabbitbrush leaves permitted high intensity combustion. Crown temperatures reached a maximum of 1,800°F. Soil temperatures were low (150°F at 1-cm, 125°F at 2-cm), probably as a result of insulation provided by the litter layer.

Only one of the four plants was sprouting in 1981. This plant was one of those that had been totally consumed. This was probably due to its isolation from surrounding fuels, which might otherwise have raised the heat load on its basal buds to lethal levels.

Horse Haven Burn 2 Results - Two individuals of green rabbitbrush were burned while in their flowering stage. The shrubs were between 40 and 50-cm tall, with 1/3-m^2 of crown area each, and litter near 2-cm deep at their bases. Foliage moisture was 66 percent.

Fire's impact on the plants consisted of 10 to 20 foot flames corresponding to 840 to 3,770 Btu/ft/sec and heat per unit area from 1,260 to 5,660 Btu/ft^2. The shrubs were not totally consumed, but foliage and stems up to 1.3-cm had been burned away. Much white ash was present (up to 10-cm), most likely as fallout from surrounding shrub consumption.

Both plants were resprouting in 1981 through 1984. Shrub height was 25-cm in 1981, 20-cm in 1982, 37-cm in 1983, and 36-cm in 1984.

Jackpot Burn 1 Results - Two green rabbitbrush plants were observed on this burn. Their maximum heights were 42-cm and 54-cm.

Foliage and stems less than 0.6-cm in diameter were consumed by the fire on both shrubs. Crown and surface temperatures were between 1,200°F and 1,500°F. Soil temperatures at 1 and 2-cm were approximately 200°F.

The plants sprouted during a year in which below normal precipitation occurred. The 42-cm plant responded with growth of 11-cm in 1981, 31-cm in 1982, 40-cm in 1983, and 36-cm in 1984. The 54-cm plant responded with growth of 8-cm in 1981, 29-cm in 1982, 26-cm in 1983, and 32-cm in 1984.

Jackpot Burn 2 Results - Three individuals of this shrub were selected for observation. Heights were approximately 50-cm and crown areas about .50-m^2. All the plants were past the flowering stage and dormant. Litter depth averaged 2-cm at their bases.

The three green rabbitbrush plants were totally consumed by the burn. One of them did not resprout. From the small sample observed, it is impossible to draw definite conclusions; however, the plant that died was surrounded by more fuel than the other plants which could have resulted in more heat impact to the basal buds. If a longer burnout occurred, there may have been greater penetration of heat into the soil. This, combined with below normal precipitation which followed the burn, could have killed the plant. The plants responded as follows: 1980 46-cm, 1981 35-cm, 1982 33-cm, 1983 36-cm, and 1984 32-cm; 1980 50-cm, 1981 25-cm, 1982 24-cm, 1983 54-cm, and 1984 47-cm. The plants reached 94 and 70 percent of their original height by 1984.

Rubber rabbitbrush (Chysothamnus nauseosus) (CHNA)

Expectations - Although rabbitbrushes are usually enhanced by fire, rubber rabbitbrush is an exception to this general response. Robertson and Cords (1957) reported no recovery of this species on two separate burns, one in California and one in Nevada. In contrast, they also record 95 percent recovery on a burn made the following year on the same date. Monsen and Christensen (1975) conclude that the intensity of the fire is important because most of the sprouting is stem sprouting, not basal or root sprouting.

Initially, a delay will occur in achene production after the burn followed by a season for peak seedling establishment. Information regarding the relation of fire intensity and postburn precipitation with plant sprouting and seed survival were not available in the literature reviewed. Since the plants were burned while in a flowering state, at a time when carbohydrate reserves are low and fire intensity was high, plant survival through stem sprouting was not expected. Seedlings are not anticipated unless seed from outside the study area is transported into the burn or soil-stored seed germinate.

Horse Haven Burn 1 Results - No plants of this species were tagged on this burn.

Horse Haven Burn 2 Results - Two rubber rabbitbrush plants were monitored for fire effects on this burn: one 60-cm tall with .13-m^2 crown area, the other 130-cm tall with .50-m^2 of crown area. Both plants had basal clumps with a diameter of 10-cm and a litter depth of 2-cm.

A fireline intensity of 3,770 Btu/ft/sec (flame lengths 20 ft) had completely consumed one plant, but left branches 1.3 and 2.6-cm in diameter on the other. The totally consumed plant was more isolated from surrounding fuels. Both plants had white ash approximately 3-cm deep about their bases.

Neither plant was resprouting in 1981. Eventhough foliage moisture was up 112 percent, the extreme fire intensity was probably responsible for this result by removal of a majority of the stemwood sprout sources. The plants failed to sprout up to 1982 and were assumed dead.

Jackpot Burn 1 and 2 Results - No individuals of rubber rabbitbrush were observed on the second burn site.

Mountain snowberry (Symphoricarpos oreophilus) (SYOR)

Expectations - Vallentine (1971) lists snowberry species as being undamaged by fire. Generally, Utah snowberry is accepted as a sprouter that may be damaged by varying fire intensities (Wright and others 1979). Pechanec (1954) showed that mountain snowberry was undamaged by fire. Depending on precipitation and effective soil moisture following the burn, snowberry plants continue to grow vigorously after resprouting, with complete recovery 15 years after burning (Pechanec and others 1954; Blaisdell 1953).

The mountain snowberry was expected to resprout and grow vigorously on the Jackpot site until sagebrush cover reestablishes dominance (Blaisdell 1953).

Horse Haven Burn 1 Results - Two mature mountain snowberry plants were observed on the first burn. They measured 50 and 89-cm in height, had crown areas averaging 1.50-m^2, and were composed of several stems growing from a tenth square meter of ground surface. The plants were in the seed dispersal stage and averaged 2-cm litter accumulation.

The crowns of these shrubs were fused with those of the surrounding big sagebrush, and despite a fairly high (81 percent) foliage and twig moisture, the mountain snowberry was consumed down to the root crown in both cases. Maximum temperatures of 1,400°F at the soil surface and 1,800°F within the canopy were generated by flames seven feet long and heat per unit area between 400 and 800 Btu/ft^2.

Beneath one plant, maximum soil temperatures reached 300°F at 1-cm and 275°F at 2-cm. White ash, 2-cm deep, was present at the base of the plants immediately after the burn.

In 1981, both plants were sprouting from the basal root crown. Shoots were over 23-cm long, indicating adequate precipitation occurred postfire (Figure 7). Browsing was evident although the site was fenced. The plants responded in the following manner: 1980 50-cm, 1981 28-cm, 1982 36-cm, 1983 54-cm, and 1984 70-cm; 1980 89-cm, 1981 23-cm, 1982 60-cm, 1983 68-cm, and 1984 66-cm. A form of leaf rust has developed on all of the snowberry leaves. This has not seemed to retard plant growth.

Horse Haven Burn 2 Results - The three snowberry plants observed on the second burn were larger (120, 120, and 90-cm; crown areas 1.5 to 3.2-m^2, respectively) and more decadent (up to 50 percent dead crown volume) than those found in burn 1. All were in a dormant vegetative stage at the time of the burn with foliage moisture at 66 percent.

Fire swept through the plant crowns at over 40 ft/min with flames 20 feet long (I = 3,770 Btu/ft/sec; Heat/Area = Btu/ft^2).

Two of the plants had foliage and stemwood up to 0.6-cm consumed, while the third individual lost only its foliage. This plant had surrounding fuel one meter away, whereas the other two plants had their crowns fused with other shrubs and grasses. White ash was observed at all the plant bases.

After one growing season, the three mountain snowberry shrubs were crown sprouting with shoots up to 40-cm long. After four years these plants were observed with shoots 61-cm long. Signs of browsing, probably by rabbits, were present. The presence of livestock or concentrations of native herbivores (such as rabbits) before and particularly after burning, can completely alter the vegetational responses to fire. Grazers exert selective pressure on certain species, making it difficult to separate changes produced by fire from those caused by grazing (Vogl 1974).

Jackpot Burn 1 Results - Two large mountain snowberry shrubs (heights 68 and 100-cm; crown areas .5 and 2.5-m^2, respectively) and one small shrub (height 33-cm; crown area .13-m^2) of snowberry were sampled. Their crowns were fused with those of surrounding big sagebrush plants, and grasses were growing up through their bases. Litter depth averaged 2-cm. The plants had shed their fruits and were in a dormant vegetative state.

The fire completely consumed each of the plants generating maximum temperatures of 1,600°F in their crowns and 1,400°F at the soil surface. Maximum sub-surface soil temperature at 1 and 2-cm was 250°F and 175°F, respectively.

All three plants were resprouting from their root crowns 1 year following the burn. By the second year, the plants described above attained 63 percent, 30 percent, and 79 Percent of their original height. The third year the plants were 66 percent, 48 percent, and 94 percent, and the fourth year these plants obtained 87 percent, 87 Percent, and 100 percent of their original height.

Jackpot Burn 2 Results - Three mountain snowberry plants ranging from 70 to 90-cm tall, with crown areas between 1 and 2-m^2 were observed. Litter depths averaged 2-cm near the plants' bases. The shrubs were in a dormant state, having shed seed.

The October burn completely consumed all three individuals, leaving black ash 2 to 3-cm deep. Each plant resprouted the following year, as expected. Below normal precipitation (Figure 6) following the burn did not hinder resprouting. By 1984 the Plants achieved 95 percent of their original height.

Utah serviceberry (Amelanchier utahensis) (AMUT)

Utah serviceberry is slightly damaged by fire and resprouts (Wright 1972; Stanton 1974). Based on observations in southern Idaho and in Utah, no serviceberry mortality was expected to occur in these burns (Wright and others 1979).

Horse Haven Burn 1 Results - Four Utah serviceberry plants were tagged preceding the August burns on the site. The plants averaged between 1 and 2-m in height, and all were in their fruiting stage.

The observations of flame length and rate of spread were 5 to 15 ft and 100 to 500 ft/min, respectively. (Intensity and heat per unit area 190 to 2,020 Btu/ft/sec and 170 and 745 Btu/ft^2, respectively.) This fire behavior, though varied in intensity, resulted in fairly uniform defoliation of the plants. This was not expected in view of the 95 percent foliage moisture content. Stemwood greater than .3-cm remained intact.

Five of the six tagged plants were resprouting one growing season after the fire. The plant that died had 2.5 times more litter (5-cm) at its base prior to burning than did the surviving plants. During the fire the litter was consumed to white ash. The plant's death may be due to the extra heating it received in the zone of its dormant basal buds, as compared to the slight charring observed at the bases of the other plants. In 1984 four of the five remaining plants were still growing vigorously reaching 54 percent of their original height. One plant seemed to be stressed and dying.

Horse Haven Burn 2 Results - In the October burn, two additional Utah serviceberry Plants were tagged. One sustained a much higher heat load than the plants in the July burn, with a heat per unit area value of 1,260 Btu/ft^2, but still resprouted. The other plant was unburned and not affected by the fire. The burned plant was observed in 1981 at 15-cm, 1982 10-cm, 1983 30-cm, and 1984 34-cm in height.

No Utah serviceberry plants were found on the Jackpot sites.

Bluegrass (Poa spp.)

Expectations - Bluegrasses are only slightly damaged by burning. Wright and Klemmedson (1965) observed no change in basal area of Sandberg bluegrass (Poa sandbergii) during any season, regardless of plant size. When plants are older and pedestalled, higher mortality can be expected (Tisdale 1959). This occurs as a result of greater buildup of dead material in older and larger plants. Fires in sagebrush communities can produce soil surface temperatures greater than 400°F which may damage Poa plants depending on the percent cover and age of overstory shrubs (Wright and Klemmedson 1965), with fire intensity generally increasing with higher fuel loading and percentage of dead material present. Harniss and Murray (1973) showed that bluegrass production dropped off 25 percent the year following a burn; but it increased steadily over the next 39 years to 140 percent of the original plot weight (values taken from a study conducted by Pechanec, Blaisdell, and Laycock 1979, unpublished).

The bluegrass component of the community was expected to survive the burns and soon to exceed preburn levels in the absence of dominant sagebrush cover.

Horse Haven Burns 1 and 2 Results - No bluegrass plants were marked or identified prior to either burn 1 or 2; but a large number were observed following the burns.

Jackpot Burns 1 and 2 Results - Only one plant of Sandberg bluegrass was marked on the first Jackpot burn. It was dormant, having already shed seed by the time it was burned.

Although the plant had its above-ground parts totally consumed along with the entire clump of shrubs within which it was growing, it resprouted the next season. Peak temperatures had reached over 1,000°F in that location, well above the 400°F lethal limit cited by Wright and Klemmedson (1965). The plant has continued to grow through 1984.

Needlegrass (Stipa spp.)

Expectations - The effect of fire on needlegrass species depends largely on the growth form and season of burn (Blaisdell 1953; Wright 1971), especially if burned during the months of June or July (Wright 1971). The very dense plant material of these grasses burns slowly and long, charring down to the growing points (Wright 1971). This longer burnout period allows subsurface charring to take place. The more dead material that is present within the bunch, the more susceptible the plant is to damage (Wright 1971). Wright and Klemmedson (1965) found needle-and-thread (Stipa comata) showing 100 percent mortality in small plants and 90 percent mortality in large plants during June burns; 20 percent overall mortality for July burns; and no mortality for August burns.

Late summer or early fall burns are less damaging because plant material becomes more tolerant of heat as tissues dry (Wright 1971). Also, when root carbohydrate reserves are lowered during the plant fruiting state, greater mortality will occur (Wright and Klemmedson 1965).

The needlegrasses were expected to survive the late summer burns because the plant tissues were dried and the seeds had set. The blackened soil should accelerate seed germination provided severe drought conditions did not follow. The productivity might be reduced in relation to unburned areas for one year after burning, but the plants were expected to recover or exceed unburned levels within 4 years (Wright and Klemmedson 1965).

Horse Haven Burn 1 Results - No needlegrass plants observed.

Horse Haven Burn 2 Results - Six needlegrass plants were monitored on this October burn, needle-and-thread, Columbia needlegrass, and Thurber's needlegrass. These were similar sized plants with basal diameters of about 8-cm; all were dormant at the time of burning. Litter was sparce about the plants, averaging less than 1-cm in depth.

Flames varied in length from 10 to 20 feet, with rates of spread of 40 ft/min. Fireline intensity was calculated to be 3,770 Btu/ft/sec and heat per unit area 5,660 Btu/ft^2. Rapid fire spread only defoliated the plants, in some cases leaving 2 or 3-cm of charred stubble. Where ash was present, it was black in color.

Five plants were observed resprouting one year after the fire. One plant was not burned and growing as before the fire. Some reasons for the recovery of the needlegrass are: the lack of accumulated dead material within the plants' bunches (these were fairly young, small individuals), the remoteness of heavy woody sage fuels and the lack of litter at the base of the plants, and perhaps most important, the plentiful rainfall that followed the burn. It should be noted that some of the plants had dead centers. The plant heights averaged 44 percent in 1981, 34 percent in 1982, 91 percent in 1983 and 85 percent in 1984.

Jackpot Burn 1 Results - One individual of Columbia needlegrass was observed. Its basal diameter was 5-cm, and the plant was dormant.

The fire consumed the plant totally, leaving black ash in its place. One year following the burn, the needlegrass was resprouting. Being located outside the exclosure, the plant showed signs of grazing. Removal of competition from sagebrush must have allowed this plant to survive despite below normal moisture available after the burn. By the fourth year the plant had grown to 160 percent of its original height.

Jackpot Burn 2 Results - Four Columbia needlegrass plants were marked on the October burn. All were dormant and averaged 5-cm of basal diameter. One plant was growing in a clump with snowberry and rabbitbrush shrubs.

Three of the four plants resprouted during the first season following the burn. The plant that died, was surrounded by highly flammable shrubs. By the fourth year the plants had basal diameters of 8, 7 and 5-cm.

Bluebunch wheatgrass (Agropyron spicatum) (AGSP)

Expectations - Bluebunch wheatgrass is slightly affected by burning. The season of burning strongly influences the degree of impact fire can have (Wright and others 1979). Plants burned in late summer and early fall show small decreases in basal area and undergo little mortality, whereas early summer burns kill 50 percent or more of the plants with large reductions in basal area (Conrad and Poulton 1966). These negative effects are usually evident in the first year following burning (Uresk and others 1976). Long-term studies (Blaisdell 1953) have observed bluebunch wheatgrass plots increasing production up to 12 years after burning, with an eventual decline after 30 years to levels near or slightly below unburned plots (Harniss and Murray 1973). Barney and Frischknecht (1974) found cover of this grass to remain constant 40 years after burning in west central Utah, until reestablishment of a juniper overstory caused a decline.

Plants of bluebunch wheatgrass were expected to survive the late August burn, with only a minor decrease in basal area. Full recovery is anticipated within three years (Blaisdell 1953; Moomaw 1957; Conrad and Poulton 1966; Uresk and others 1976).

Horse Haven Burn 1 Results - Three individuals of bluebunch wheatgrass were tagged Prior to the August burn at Horse Haven. Seed dispersal had been completed and the plants were in a dormant state.

Flames of 6 to 8 feet, moving at 25 to 200 feet per minute, burned in the vicinity of these plants. In terms of fireline intensity and heat per unit area, this fire behavior represents 280 to 500 Btu/ft/sec and 155 to 660 Btu/ft^2, respectively.

All the plants had only charred tufts remaining after the fire, but as expected, all were resprouting the following year. Two of the plants; however, showed a reduction in live basal area of between 25 and 75 percent.

Horse Haven Burn 2 Results - No bluebunch wheatgrass plants were observed.

Jackpot Burn 1 Results - Two bluebunch wheatgrass plants were monitored on this burn. They had basal diameters of 7-cm and had already shed seed. Both individuals were located with clumps of several shrubs and other grasses, all of which were in contact with each other. Litter was approximately 2-cm deep and pedestalling was not indicated for either plant.

The late August burn completely consumed both plants down to white ash of 2-cm depth. Maximum temperatures reached 2,000°F within the crowns of the plants in the clump, dropping to 1,300°F at the base of one bluebunch wheatgrass plant. Peak soil temperatures beneath this plant were 200°F at 1-cm and 175°F at 2-cm.

One year after the fire, one plant was resprouting and the other dead. One possible explanation is that the surviving plant had its basal buds isolated from prolonged heating, even though the (1,300°F) peak temperature was quite high. Also, it was noted from the preburn photographs that the dead plant had been surrounded closely by several rabbitbrush plants, which have a high resin content that burns intensely. Basal area of the surviving plant increased from .5 x .5-cm in 1981 to 9 x 12-cm in 1984.

Jackpot Burn 2 Results - Five individuals were tagged on this burn, each averaging 8-cm in diameter. All these plants were beneath sagebursh canopies and had much dead grass foliage and other plant debris surrounding them. The plants had shed seed and were dormant at the time of this October burn.

Every bluebunch wheatgrass plant was totally consumed by the fire. Because of the shading of the fine fuels around the plants, their moisture contents were probably a little higher than exposed material. Hence, little white ash was observed, and most of the remains at the plant bases were only charred.

Unexpectedly, four of the five plants were dead the following season. The one surviving plant was farther from heavy accumulations of litter. By the second year it was 65 percent of its preburn height and by the fourth year 127 percent of its original height. The basal area, although initially reduced, returned to its preburn dimension. The more moist conditions beneath the sage crowns, though possibly reducing the fire intensity, may have rendered these plants more susceptible to damage by raising the heat conductivity of the plant tissues (Wright and Klemmedson 1965). The lack of ample postfire precipitation (Figure 6) may also have reduced the wheatgrasses' chance for survival.

Bottlebrush squirreltail (Sitanion hystrix) (SIHY)

Expectations - Bottlebrush squirreltail was expected to sustain some reduction of basal area in the July burn and undergo slight damage in the October burn; in both cases ultimately surviving the fires. The damage resistance of bottlebrush squirreltail can be attributed to its low density of dead plant material within the bunch (Wright and Klemmedson 1965; Wright 1971). As a result, the aerial plant parts burn quickly, with a minimum of heat penetration to the growing points. Other workers have found no mortality in squirreltail plants burned in mid-June and October (48 percent basal area reduction), but plants burned in mid-May in a drought year suffered 30 percent mortality (73 percent basal area reduction) (Wright and others 1979).

Horse Haven Burn 1 Results - Four plants were marked for observation in the late August burn. The plants were in the seed dispersal stage and averaged 12-cm basal diameter. The height ranged from 30 to 50-cm. Litter accumulations at the bases of the plants were from 0.5 to 2-cm in depth.

The fire burned the plants with an intensity of 200 Btu/ft/sec (5 ft flames), generating heats per unit area from 90 to 660 Btu/ft. Surface temperature was recorded at over 1,000°F at the bases of two plants, but no white ash was observed, only charring of the bunchgrass tufts.

The second year two plants died and were replaced by cheatgrass; one plant was severely damaged; and the fourth plant, which was pedestalled, recovered to 50 percent of its original height. By the fourth year, plant height was 200 percent of the original. The basal diameter was unchanged. Grazing pressure was evident on some of the plants.

Horse Haven Burn 2 Results - In the October burn, two bottlebrush squirreltail plants were marked for examination. Their basal diameters were 10 and 13-cm, with heights of 40 and 60-cm. These plants had slightly more dead material within their bunches than those of the July burn.

Fire behavior was much more intense on this burn, with fireline intensities of 3,770 Btu/ft/sec (20 ft flames) and heat per unit area of 5,660 Btu/ft^2. Nevertheless, the plants were only charred down to the basal tuft, and proceeded to sprout after one season, though they had less herbage than before the fire.

The survival of the squirreltail plants in both fires is probably the result of the fast fire rate of spread (20+ feet per minute) through the plants; as well as the above normal rainfall which followed in the autumn (See Figure 7). Both plants developed dead centers, however. The recovery was slow and in 1984 the plants were approximately 50 percent of the original height and suffered a reduction in basal area by half.

Jackpot Burns 1 and 2 Results - No individuals of this bunchgrass were found.

Basin wildrye (Elymus cinereus) (ELCI)

Expectations - Basin wildrye is only slightly damaged by fall fires (Vallentine 1971). Older plants with large proportions of dead material within the perennial crown can be expected to suffer higher mortality than younger plants having little debris. The coarseness of basin wildrye foliage also resists prolonged burning, hence the plants avoid exposure to sustained heating of their basal growing points. High soil moisture at the time of burning is harmful because it increases the conduction of heat from the surface to the root system. If the wildrye burns late in the season, after having gone dormant, it is more likely to survive and resprout than at any other period in its growth cycle.

Wildrye plants were expected to survive the late season burns; and if followed by adequate rainfall, to resprout the grow equal to or greater than preburn rates. Three to four years following the burn; however, this grass may decline in vigor and frequency (Young and Evans 1978).

Horse Haven Burn 1 Results - One plant of wildrye was tagged for observation. The plant was 25-cm in basal diameter with a height of 110-cm, and dormant at burning time. Litter was accumulated to a depth of 6-cm surrounding its base.

Six foot flames (280 Btu/ft/sec) with a rate of spread of 25 ft/min (660 Btu/ft^2) defoliated the plant, leaving charred stubble. The maximum temperature recorded within the basal crown was 1,500°F.

Still, the wildrye plant had resprouted by July of the next season, although it developed a dead center after two years. Dry soil (6 percent taken at 15-cm depth) before the burn, combined with sufficient rainfall (Figure 7) after the burn may have contributed to the plant's survival. The plant grew from its preburn height of 110-cm to 105-cm in 1981, 91-cm in 1982, 129-cm in 1983 to 77-cm in 1984. Signs of grazing were present during all years.

Horse Haven Burn 2 Results - Two basin wildrye plants were monitored. Basal diameters were 65 and 49-cm, with litter averaging a depth of 3-cm. Both plants were in a dormant state when burned.

A fire many times more intense than that in Burn 1 only defoliated the plants. Flames were 20 feet long (3,770 Btu/ft/sec), rate of spread was 40 ft/min, and heat per unit area was 5,660 Btu/ft^2. Blackened stems and leaves measuring 12-cm long remained over the base area of the plants.

Both plants survived the burn, as re-sprouting was observed the following season. By the second year, one plant was sprouting from five areas on the old crown and attained 50 percent of its original height, but had a reduced basal area. The second plant was 100 percent of its original height, had a slightly reduced basal area, and no dead center. By the fourth year the plants were at 97 percent of the original basal area and had a height of 84 percent the original.

Jackpot Burn 1 Results - The one basin wildrye monitored, died.

Jackpot Burn 2 Results - Only one basin wildrye was monitored. The fire initially burned the coarse, dead stem and decreased the basal diameter; but by the second year postburn the base was larger than its preburn dimensions. The plant was 73 percent of its preburn height. By year four, the plant nearly doubled its basal diameter by 100 Percent and was 142 percent of the original height.

Forbs (general)

Expectations - Many forbs, especially the rhizomatous ones, make rapid recovery after burning and produce an increased amount of herbage within three years. Others, particularly those with woody bases, are slower to recover. None of the perennial forbs are permanently damaged, and many apparently benefit from burning (Blaisdell 1953). Klebenow and others (1977) found that increases in forb cover were the most important aspect of succession following burning.

Forbs that are dormant at the time of burning are not harmed; whereas, forbs actively growing or flowering are susceptible to damage (Britton 1977). Most forbs have set seed and are disintegrated by the fall, which is why burning is least harmful at that time. Pechanec and Stewart in 1944 classified forbs according to their susceptibility to fire. In a later study, Pechanec and others (1954) observed that forb species that spread by rootstocks increase more rapidly after burns than those reproducing by seeds alone. In the latter group, such species as arrowleaf balsamroot and tailcup lupine were expected to make an initial flush after the fire, drawing on previously stored seeds. Further increases; however, would have to await more seed production.

Blaisdell and others (1953) saw forbs one year after a fire increase in density and biomass 30 to 60 percent over unburned areas, and increase 30 to 100 percent after 3 years. (Their study was conducted on the Snake River Plains, Idaho, a site similar to the Jackpot site.) Twelve years after the burn, total forb yield on the burned plots was still higher than on the unburned plots. But 30 years later, Harniss and Murray (1973) showed that these trends were reversed as sagebrush gradually returned to dominate the site.

Blaisdell (1953) also noted an inverse relationship between forb production and fire intensity. High intensity burns had lower forb production. This was probably the result of the fire's destruction of stored seeds and damaged to rootstocks of dormant plants.

No assessment of forb production was made in this study, but changes in relative cover are assumed to reflect closely the changes of forb density and production. Given the fall burns, and the fact that all the forbs were dry, it was expected that an initial flush of forb cover would immediately follow the burns, with total forb cover and number of species markedly greater than before the burn. Beyond the first year, decline of the forb component is anticipated, the rate depending on how fast sagebrush, rabbitbrush, cheatgrass, and other invader species reoccupy the site.

Horse Haven Burns 1 and 2 Results - Only a trace of forb species was observed before both burns; therefore, no individual plants were monitored. Pricklypear (Opuntia polyacantha) a species of cactus found on the preburn site, was not observed two years postburn.

Such an explosion of species diversity may be partially due to the plentiful rainfall that followed the burns (Figure 7). Stickseed (Hackelia spp.) had the highest forb cover on the site, with wayside gromwell, milkvetch, lupine, blue-eyed Mary (Collinsia parvifolia), fiddleneck (Amsinckia spp.), tansymustard (Descurainia), povertyweed (Iva axillaris), and tapertip hawksbeard (Crepis acuminata) also tallied.

Jackpot Burns 1 and 2 Results - In burn 1, two individuals of mint (Agastache urticifolia), and one each of lupine and arrowleaf balsamroot were tagged; in burn 2, only two mints were marked.

Observations of these plants immediately after burn 1 indicated that both mint plants had all of their above-ground parts completely consumed. This was also true of the balsamroot and lupine plants. Maximum surface temperatures on these individuals were recorded at 1,200°F to 1,500°F. Soil temperatures reached peaks of 250°F and 175°F at 1 and 2-cm below the surface, respectively. Little can be said regarding the linkage of surface peak temperature with ash conditions, since half the cases of 1,200°F temperatures had black ash remaining, and on the other half white and grey ashes were present.

Both mint plants were totally consumed by the second burn, leaving a residue of white ash.

Resampling both burn sites in 1981 showed that the balsamroot plant was killed by the August burn, probably as a result of the burnout of adjacent woody sage fuels. The lupine and mint plants resprouted, with lupine being the most abundant forb on the site as a whole. Only one of the two mint plants survived the October burn, perhaps because it was more isolated from surrounding fuels than the plant that died. However, this site was not fenced, and the sprouting plant may have been eaten by cattle or deer. Complete precipitation records for this site are lacking, but general estimates indicate that rainfall levels were slightly below normal following the burn, accounting partly for the relatively low forb species diversity on the site.

The forb component on Jackpot-1 increased dramatically from 1.2 preburn to 25.3 percent four years postburn. Jackpot-2 also saw an increase from a trace of forbs to 39.1 percent. At Horse Haven forb response was from 0 percent to 5.8 percent and 0 percent to 5.4 percent for burns 1 and 2, respectively.

Summary

As anticipated all of the big sagebrush plants died and seeding reestablishment has started between three to four years postburn. The antelope bitterbrush plants also suffered a high mortality. Several plants that were on the perimeter of the burn survived. In these areas, the rate of spread was slow and the flame lengths were between 4-6 feet.

The green rabbitbrush plants that were not sprouting were probably exposed to a very high heat per unit area. The rubber rabbitbrush plants did respond according to the literature. All of these plants died.

All of the mountain snowberry plants resprouted as expected. The Utah serviceberry plants suffered slightly. This was anticipated according to the literature.

Bluegrass plants survived on all of the sites and the needlegrass species suffered a slight decrease. The bluebunch wheatgrass species seemed to be more sensitive to late fall burns as several plants died. The bottlebrush squirreltail plants survived the late summer burns but suffered a 50 percent mortality during the late fall burns being replaced by cheatgrass. On all burns the basin wildrye survived contrary to the literature predicting mortality during the late summer burns.

This study has set the standards for monitoring to be followed in Nevada. On each prescribed burn key plants will be tagged, photographed and measured. Each plant will be remeasured following the burn for several years.

It is hoped that this site and plant response data will be stored on a computer program to create a catalogue for later reference.

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Fire as a Management Tool in Southeast Idaho--A Case Study

O'dell A. Frandsen

Abstract

Prescribed burning is becoming an accepted land treatment method. Bureau of Land Management policy allows use of the method and requires prescribed fire planning prior to burning.

Prescribed burns in sagebrush grassland areas north and west of St. Anthony, Idaho, significantly improved vegetation composition for big game. Shrubs decreased from 65 percent to 25 percent, grasses increased from 18 percent to 25 percent, and forbs increased from 17 percent to 22 percent. The land treatment enhanced desired plant growth, minimized detrimental affects to the land, and was cost-effective.

Introduction

In recent years a growing number of individuals from the public and professionals of natural resource management have begun to accept and support the concept of fire as a land management tool. Historically, wide acceptance was slow despite claims by early pioneers of range science and plant ecology that under proper conditions burning is beneficial to the range. The vast majority of the public held the view that all fires are hazardous and should be suppressed as quickly as possible at all costs. More and more, however, professionals and interested non-professionals alike are beginning to see the beneficial results of fires under controlled conditions.

During the past decade the Bureau of Land Management (BLM), in keeping with the recent trend, has made considerable changes in its fire management policies. No longer does the BLM hold to its old belief that its task in fire management is solely to suppress fires. It now recognizes fire as a valuable resource management tool that can be used to enhance wildlife habitat and improve range conditions. In addition to the environmental considerations, it has come to recognize that in many cases fire is the most cost-effective land treatment method available (Figure 1). Specifically, Bureau policy has changed to allow limited suppression plans where the control of fire is extremely difficult and/or where the resource values do not warrant the expense of usual suppression activities. The policy allows managers to use fire as a management tool and requires them to prepare a prescribed fire plan in advance of natural or intentional ignition.

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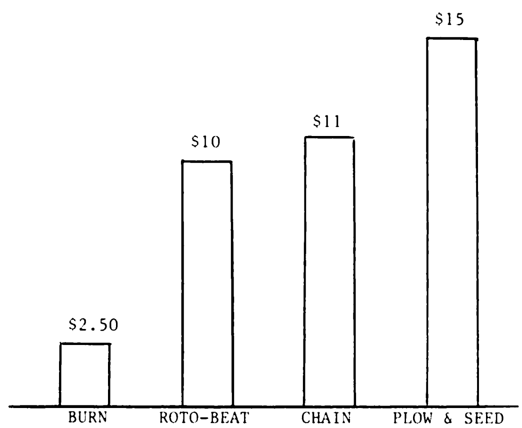
Figure 1. Land Treatment Costs Per Acre.



The fire suppression program in the Idaho Falls District, which encompasses 2.5 million acres of public land in southeast Idaho, is one of the larger programs in the Bureau. During the last 7 years an average of 75 fires occurred burning 20,500 acres per year at an average of 300 acres per fire. The District's first prescribed burn plan was developed in 1978. Initially, the plan was hampered by lack of commitment at all levels of the Bureau, by lack of expertise at the District level and by inadequate land use planning. The District largely overcame these problems by training a District multi-disciplinary team of both line and staff personnel.

Three primary factors helped the District move into its fire management program. Firstly, the completion of the Sands Habitat Management Plan (HMP) in 1978 provided the basis for the program. This plan, which includes 200,000 BLM acres, 85,000 State acres, 100,000 private acres and 20,000 State Fish and Game acres, emphasizes the manipulation of vegetation to improve elk habitat. During its development it became apparent that controlled burning was the primary tool needed to achieve the plan objectives. The second major factor was the issuance of the Bureau's Interim Management Guidelines for Wilderness Areas in 1979. These guidelines direct BLM to begin planning to protect sensitive areas, such as the Sands HMP area, from unnecessary damage created by fire suppression activities. The third factor was the completion of the District's Big Desert Planning Unit Environmental Impact Statement (EIS) in 1981. This EIS paved the way for development of fire management plans within a one million-acre area of the District.

From 1979 to 1981, the District's prescribed burning program was centered in the Sands HMP area north and west of St. Anthony, Idaho. Vegetation in the District, which has elevations from 4,500 feet to 8,000 feet, ranges from typical cold desert varieties of grass, sagebrush and juniper at the lower elevations to fir and pine at the higher elevations. The prescribed burns were located primarily in the sagebrush grassland community. They were designed to remove the competition of sagebrush and to enhance the growth of the other more desirable shrubs, grasses and forbs. The principle species within the HMP area are listed below:

Shrub

big sagebrush (*Artemisia tridentata*)
antelope bitterbrush (*Purshia tridentata*)
common chokecherry (*Prunus virginiana*)
rabbitbrush (*Chrysothamnus* spp.)
mountain snowberry (*Symphoricarpos oreophilus*)

Grass

needle-and-thread (*Stipa comata*)
blue grass (*Poa* spp.)
fescues (*Festuca* spp.)
Indian rice grass (*Oryzopsis hymenoides*)

Forb

arrowleaf balsomroot (*Balsamorhiza sagittata*)
mule's-ear wyethia (*Wyethia amplexicaulis*)
mullein (*Verbascum* spp.)
lupines (*Lupinus* spp.)
sticky geranium (*Geranium viscosissimum*)
common buckwheat (*Eriogonum* spp.)
larkspur (*Delphium* spp.)
foothill death camas (*Zygadenus paniculatus*)

Under the Sands HMP, the primary objective is to increase forage for wildlife that use the area during spring, fall and winter. Inhabiting wildlife includes 2,000 elk, 1,400 deer, 5,000 antelope and 100 moose. The burned areas allow the wildlife to stay at higher elevations two to three weeks longer in fall and spring while providing additional forage for both wildlife and livestock. The prescription used for the HMP burns evolved from early trial and error efforts based on an established formula. The resulting prescription, designed so that only 50 percent of the area is burned, is shown below:

Prescription Parameters	Units	Range
Burning Index	Flame height in tenths of feet	30-45
Windspeed	Miles Per Hour	5-18
Temperature	Degrees Fahrenheit	65-75
Relative Humidity	Percent	12-23
Soil Moisture	Percent	50

The growth stage of the vegetation is a very important aspect of the prescription. District personnel encountered major problems during early attempts at burning because they were trying to burn at the wrong time of year. They soon discovered that the vegetation must be cured and dormant for best results. This places less stress on the plants and achieves better recovery. At times, they learned it is necessary to wait for a hard freeze to dry out some of the finer fuels. They conduct their burns in the fall, usually September and October. Most often, the burns were covered with snow within two or three weeks after the burn. This has proved to have a positive effect on vegetation recovery, particularly bitterbrush. In the Sands HMP area, positive changes in vegetation composition after burning at the right time of year have been dramatic and are shown below:

Vegetation Type	Composition Before Burn	Composition After Burn
Shrubs	65%	25%
Grasses	18%	25%
Forbs	17%	22%

After burning, studies consistently indicated a significant increase in grass production. The amount has varied by year, but even in drought years production has been from two to three times greater than in unburned areas.

Some interesting changes to bitterbrush were discovered. A general reduction in plant numbers occurred, ranging from a high of 50 percent to a low of 10 percent with an average reduction of 30 percent. The extent of reduction was controlled proportionate to the intensity of the fire. Studies show that even burns removing up to 30 percent of the plants had an insignificant negative impact since many of those lost were unimportant to wildlife because they were either decadent or unavailable for wildlife consumption. In areas where any loss of bitterbrush would be unacceptable, spring burning should be considered. District surveys indicate high bitterbrush losses from summer burns. Although most plants resprouted in the fall, few survived the winter. In all the District's prescribed burns, however, a good increase in leader growth occurred. The average leader growth of bitterbrush on burned and unburned areas is shown below:

Years After Burn	Burned Area	Unburned Area
1	10.58 cm	2.97 cm
3	3.43 cm	2.12 cm

In summary, prescribed burns conducted at the right time of year have resulted in a significant increase of native grasses and forbs. Prescribed fire has provided the land manager a cost effective tool that has enhanced desired plant growth and minimized detrimental affects to the land. Given the following special considerations and precautions, this tool, from all indications, has excellent potential for future land treatment applications:

1. Design a large enough burn to avoid grazing problems.

2. Do not permit livestock grazing for two years after burning.

3. Area must have ample composition of desirable species.

4. Burn must stay within the prescription.

5. Immediately suppress a prescription should it exceed the threshold limits of the prescription.

6. Do professional work.

Early Effects of a Fall Burn in a Western Wyoming Mountain Big Sagebrush-Grass Community

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Abstract

A high-elevation mountain big sagebrush (Artemisia tridentata ssp. vaseyana (Rydb.) Beetle) - grass plant community was burned in September 1983 and evaluated one year later. Mountain big sagebrush cover was reduced to zero on 40% of the west slope and 60% of the east slope for an overall reduction of 50%. Grass production and gray horsebrush (Tetradymia canescens DC.) cover were similar on burned and unburned areas of both slopes. Forb production and Douglas rabbitbrush (Chrysothamnus viscidiflorus (Hook.) Nutt.) cover were greatest on burned areas of the west slope, and mountain big sagebrush seedling density was greatest on burned areas of the east slope.

Introduction

Big sagebrush (Artemisia tridentata Nutt.) - grass communities are extensive and important in western Wyoming. Many of these plant communities have a large proportion of sagebrush cover (40+%) with a diverse understory. This provides land managers with the opportunity to alter community structure via chemical, mechanical, and/or burn treatments to improve livestock forage production and availability.

Succession following wildfires in Wyoming indicates that most grasses, forbs, and browse, except big sagebrush, respond positively to burning. The literature indicates that the response can be improved by prescribed burning under optimum conditions. Further, prescribed burning, where feasible, is often cheaper and more compatible with land management objectives than chemical or mechanical treatments. Therefore, the BLM Rock Springs District has initiated a prescribed burning program to reduce sagebrush cover, increase livestock forage production, and increase forage availability. This paper summarizes the first-year vegetation response to a fall burn in a mountain big sagebrush (A. t. ssp. vaseyana (Rydb.) Beetle) - grass community in western Wyoming.

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Site and Burn Description

About 480 ha (conversions to English units are provided in the Appendix) in the Rock Creek Allotment, Kemmerer Resource Area, were burned in September, 1983. Burn objectives were to reduce big sagebrush cover by 50 to 75% in a mosaic pattern, and to double forage production. The site is on east and west slopes of a north-south ridge, at 2200 to 2300 m elevation, about 20 km northwest of Kemmerer, Wyoming. Soils on the west slope are loamy-skeletal, mixed, Argic Cryoborolls derived from calcareous quartzite. Soils on the east slope are intermingled, consisting primarily of loamy-skeletal, Calcic Cryoborolls (40%), fine-loamy, carbonatic, Argic Cryoborolls (25%), and loamy-skeletal, Lithic Cryoborolls (15%) derived from limestone. The west slope contains rounded quartzite fragments through the profile, and the east slope contains angular limestone fragments, often on the surface (personal communication with Chester Novak, BLM soil scientist, Kemmerer).

Slope averages 25% on both aspects. Average annual precipitation at Kemmerer (elevation 2121 m) is 230 mm although precipitation was 254% of average during 1983 and 169% of average during the first eight months of 1984. Also, precipitation was 527% of average during August 1983, and 407% of average during September 1983, the month of the burn (NOAA 1983, 1984).

The site supported a mountain big sagebrush-grass community. The shrub layer was dominated by mountain big sagebrush with antelope bitterbrush (Purshia tridentata (Pursh) DC.), alkali sagebrush (A. longiloba (Osterh.) Beetle), mountain snowberry (Symphoricarpos oreophilus Gray), Douglas rabbitbrush (Chrysothamnus viscidiflorus (Hook) Nutt.), gray horsebrush (Tetradymia canescens DC.), and Utah serviceberry (Amelanchier utahensis Koehne) also present. Common forbs were sulfur buckwheat (Eriogonum unbellatum Torr.), flowery phlox (Phlox multiflora A. Nels.), arrowleaf balsamroot (Balsamorhiza sagittata (Pursh) Nutt.), hollyleaf clover (Trifloium gymnocarpon Nutt.), lambstoungue groundsel (Senecio intergerrimus Nutt.), oblongleaf bluebells (Mertensia oblongifolia Nutt.), tansy mustard (Descurania sp. Webb & Berth.), silky lupine (Lupinus sericeus Pursh), bastard toadflax (Commandra umbellata (L.) Nutt.), western yarrow (Achillea millefolium L.), and bushy birdsbeak (Cordylanthus ramosus Nutt. ex Benth. in DC.).

The grass layer included Sandberg bluegrass (*Poa sandbergii* Vasey), Letterman needlegrass (*Stipa lettermanni* Vasey), bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith), thickspike wheatgrass (*A. dasystachyum* (Hook.) Scribn.), prairie junegrass (*Koleria cristata* (L) Pers.), and Great Basin wildrye (*Elymus cinereus* (Scribn. & Merr.).

The east slope was burned on 23 September 1983 by strip headfiring with a helitorch in a downslope wind of 0 to 16 km hr^{-1}. Ignition began at 1330 and ended at 1830 MDT. Dry bulb temperature was 19 to 17°C, relative humidity 49 to 46%, and 10-hr fuel stick moisture content increased from 18 to 23%. Cumulus cloud cover was 40% at ignition and decreased to 20%. A trace of rain fell about 1500 on the day of the burn and again three days later.

The west slope was burned on 30 September 1983 by firing upslope, with drip torches, into a light and variable wind that occasionally gusted to 30 km hr^{-1}. Ignition began at 1230 and ended at 1630 MDT. Dry bulb temperature at ignition was 18°C, relative humidity 40%, and 10-hr fuel stick moisture content was 20%. Cloud cover was 40%. Additional weather observations were not recorded, but about 1500 a windshift occurred with a noticeable increase in relative humidity.

Methods

The west and east slopes of the burn were in different pastures, appeared to have different management histories, differed in aspect, and were ignited differently. Plant community composition also appeared different (e.g., antelope bitterbrush conspicuous on the east but not on the west slope). Therefore, the west and east slopes were evaluated separately in mid-September 1984. Specific sampling objectives were to determine if cover and current herbaceous standing crop were different between burned and unburned areas and between slopes, to determine if burning affected mountain big sagebrush germination on each slope and to determine if objectives of the burn had been accomplished. Also, because populations of gray horsebrush and Douglas rabbitbrush were present, these species were recorded separately to monitor response to burning.

On each slope, three points were randomly located in burned areas that were adjacent to unburned areas. From each point, a 30 m transect was established into the burn. Plant cover was estimated along each transect using the line interception method (Canfield 1941, Kaiser 1983) for the following plant categories: grasses, forbs, mountain big sagebrush, gray horsebrush, Douglas rabbitbrush, and other shrubs. Intercepted length of the bases of grasses and single stemmed forbs, and canopy cover of rosette forbs and shrubs, were measured and recorded to the nearest 0.25 cm. Mountain big sagebrush seedlings in a 60 m^2 area transect, superimposed over the line transect axis, were then counted to estimate big sagebrush seedling density. Similar line and area transects were completed in adjacent unburned areas, and the adjacent area transects were treated as paired plots. This scheme resulted in 12 transects, with three each in burned and adjacent unburned areas on each slope. In addition, current herbaceous standing crop in forty, 0.45 m^2 rectangular quadrats was clipped to within 1 cm of the soil surface, separated as grasses or forbs, oven-dried at 60°C to constant weight, then weighed to the nearest g to estimate current standing crop. Analysis of variance and Duncan's New Multiple Range Test (Duncan 1955) were used to separate means of current herbaceous standing crop and cover on burned and unburned areas of the east and west slopes. The t test for paired plots (Little and Hills 1978) was used to contrast mountain big sagebrush seedling density between burned and unburned areas of the east and west slopes.

Results and Discussion

Sagebrush Seedling Density

Due to variation among area transects, big sagebrush seedling density (Table 1) could not be statistically separated between burned and unburned areas on the west slope. On the east slope, however, there were significantly (P=.05) more seedlings on the burned areas than on the unburned areas. The reasons for these differences are unclear; however, several possibilities exist. First, the east side was ignited with a helitorch, resulting in a more intense fire. Since heat stimulates germination of mountain big sagebrush seeds (personal communication with Dr. A. H. Winward, USFS), the ignition technique may have contributed to big sagebrush seedling establishment on the east slope. Second, grass cover was somewhat less on the east slope, thus providing more opportunity for sagebrush seed germination and survival. Third, the east side may be inherently different than the west side with respect to soil moisture conditions and insolation. Finally, this was a first-year response and seedling density may change over the next several years.

Table 1. Big sagebrush seedling density (No. ha^{-1}) one year after burning a mountain big sagebrush-grass community.

	West Slope[1]	East Slope[1]
Burned	299^a	2691^a
Unburned	1255^a	240^b

[1] Means within columns followed by the same letter are not different (P=.05).

Herbaceous Production

Herbaceous production in sagebrush-grass communities normally decreases the first year following burning, then increases above preburn levels in subsequent years (Britton and Ralphs 1979). Based on current standing herbaceous standing crop (Table 2), first-year reduction did not occur on the Rock Creek burn. Total herbaceous standing crop was significantly (P=.05) greater on burned areas of the west slope than on adjacent unburned areas, and was greater than burned or unburned areas of the east slope. The difference, however, was due to increased forb production, primarily western yarrow and tansy mustard. Although not significant, the trend suggested that grass production was also greater on burned areas of the west slope, and that the west slope may have been more productive than the east slope.

The standing crop response was not unexpected because neither the dominant grass, thickspike wheatgrass, nor the dominant responding forb, western yarrow, are damaged by fire (Wright et al. 1979). These data suggest that the current policy of mandatory two season grazing deferment following burning may not be warranted in some cases, especially if precipitation prior to, and following, the burn are substantially above average. Fall and spring precipitation have been shown to be the most important variables affecting grass production on Wyoming rangelands (Ries 1973).

Table 2. Current herbaceous standing crop (kg ha[1]) one year after burning a mountain big sagebrush grass community.

Area[1]	Grasses	Forbs	Total
West Slope Burned	270[a]	227[a]	497[a]
West Slope Unburned	201[a]	113[b]	414[b]
East Slope Burned	162[a]	134[b]	296[b]
East Slope Unburned	158[a]	112[b]	270[b]

1 Means within columns followed by the same letter are not different (P=.05).

Cover

Mountain big sagebrush cover was reduced to zero (Table 3) on about 40% of the west slope and 60% of the east slope, thus accomplishing the first objective of the burn; the mosaic was better on the west side, probably because better control was possible with drip torches. Two less desirable sprouting shrubs, gray horsebrush and Douglas rabbitbrush, responded differently. Gray horsebrush cover decreased on burned areas of the west slope but remained unchanged on the east slope. This follows the initial pattern reported in Idaho and summarized by Wright and Bailey (1982); however, on the Idaho burn, horsebrush doubled after three years, increased five-fold after 12 years, and was still 60% above the unburned control after 30 years. Although horsebrush is not a preferred shrub, increases of two- or three-fold from current low levels on the Rock Creek burn should not detract from further burn plans because the undesirable nature of horsebrush is based, in part, on its toxicity to sheep rather than cattle (Kingsbury 1964). Further, gray horsebrush is less toxic than other horsebrush species, is unpalatable to cattle, and must be consumed with black sagebrush for toxicity to occur (Johnson 1978).

Douglas rabbitbrush appeared to increase on both slopes although the increase was significant (P=.05) only on the west slope. This response is different than on the Idaho burn where rabbitbrush was reduced 59% the first year, then doubled after three years and tripled after 12 years (Wright and Bailey 1982). If data from Idaho are applicable to the Rock Creek burn, rabbitbrush will probably dominate both slopes for at least 15 years. This, plus speculation that mountain big sagebrush will reach preburn levels in 20 to 30 years, suggests that the Rock Creek site should not be reburned for at least 15 years, and probably not for 20 to 25 years. The reproductive as well as vegetative release of rabbitbrush was interesting on the Rock Creek site; almost no Douglas rabbitbrush plants flowered in the unburned areas whereas almost all plants in burned areas flowered.

Table 3. Cover (%) of grasses, forbs, and shrubs one year after burning a mountain big sagebrush-grass community in western Wyoming.

	Vegetation Component[1]							
	grass	forbs	Artrv[2]	Teca[2]	Chvi[2]	OS[2]	TS[2]	TV[2]
West Slope								
Burned	2.08[a]	3.45[a]	0[a]	0.03[a]	6.36[a]	0.13[a]	6.53[a]	12.06[a]
Unburned	0.93[b]	1.09[a]	30.87[b]	0.84[b]	2.01[b]	6.12[b]	39.84[b]	41.87[b]
East Slope								
Burned	1.33[b]	7.77[a]	0[a]	0.07[a]	3.56[b]	0.64[a]	4.28[a]	13.38[a]
Unburned	0.52[b]	1.05[a]	40.45[b]	0.01[a]	1.57[b]	3.34[a]	45.36[b]	46.93[b]

1 Means within columns followed by the same letter are not different (P=.05).

2 Artrv=mountain big sagebrush, Teca=gray horsebrush, Chvi=Douglas rabbitbrush, OS=other shrubs, TS=total shrubs, TV=total vegetation.

Grass response is open to speculation. Cover data (Table 3) suggest that grass cover on burned sites was double that on adjacent unburned sites. Standing crop data (Table 2), however, indicate that burned and unburned areas produced similar amounts of grass on both slopes. This apparent contradiction may be an artifact of the sampling methods. However, one reasonable explanation is that fewer plants in unburned areas produced more phytomass per plant due to shade-induced etoliation. Alternately, plants in burned areas may have produced less phytomass per plant due to burn damage. In any event, abnormally large amounts of precipitation preceding and following the burn probably ameliorated the first-year depression normally reported for grass in sagebrush-grass communities.

Summary

(1) Mountain big sagebrush cover was reduced to zero on 50% of the burn site, thus satisfying one objective of the burn. Forage production is expected to increase but whether the second burn objective is met remains to be seen.

(2) Grass production on burned areas equaled or exceeded that on unburned areas. This first-year response is different from results of most burns and may be attributable to above-average precipitation before and after the burn.

(3) Gray horsebrush cover was similar between burned and unburned areas of both slopes.

(4) Douglas rabbitbrush cover on burned areas of the west slope was greater than on unburned areas of either slope.

(5) Big sagebrush seedling density on burned areas was greater than on unburned areas of the east slope but not of the west slope.

Recommendations

(1) Grasses are recovering more quickly than anticipated, and two seasons' nonuse may not be necessary for full recovery. However, excessive livestock use, or use early in the spring while the soil is still moist, should be avoided.

(2) The site should be reevaluated in fall, 1986 to track grass and forb production, sagebrush reestablishment, and cover of gray horsebrush and Douglas rabbitbrush.

(3) Research is needed to determine the optimum post-burn rest periods that are necessary for grass recovery on high elevation sites. These sites appear to recover quickly, especially when precipitation is ample, and may not require the same rest period as low elevation sites.

(4) Research is also needed on the best combination, timing, and order of treatments to reduce mountain big sagebrush, gray horsebrush, and Douglas rabbitbrush where they occur together on high elevation sites.

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Appendix. Factors (approximate) to convert selected SI Units to English Units.

SI Unit	multiply by	to obtain English Units
cm	0.3937	inches
m	3.2808	feet
km	0.6214	miles
ha	2.4710	acres
°C	1.8°C+32	°F
km hr^{-1}	0.6214	miles $hour^{-1}$
kg ha^{-1}	0.8922	pounds $acre^{-1}$

Can Reason Suppress the Fire Demon?

Robert G. Lee

Abstract

Modern fire management is typified by an unprecedented reliance on rational thought and careful analysis. This paper examines the historical uses of the demon as a symbol for wildfire to show the novelty of a rational approach. The use of mythical thinking involving a fire demon is contrasted with modern scientific rationality and the democratic purposes which guide the use of reason. The need to avoid new crusades that can be just as irrational is emphasized.

Introduction

Modern fire management is characterized by an unprecedented emphasis on scientific rationality and technical expertise. Fire management prescriptions are derived from knowledge of the behavior, ecology, and effects of fire in particular ecosystems. Even economic, social, and political considerations have been subjected to rational analysis for purposes of developing better fire management prescriptions. As a result of relatively rapid progress in transforming fire control into fire management, scientists and practitioners have underemphasized the novelty of their new enterprise. They seldom realize how unusual they are in basing resource management practices on facts generated by carefully reasoned analysis. Correspondingly, they are often unaware of the vigilance required to maintain such a rational and analytical approach to policy-making and management planning.

Rationality in fire management contrasts sharply with the use of mystical symbolism in traditional fire control. I have spent many years attempting to understand why scientific and technical rationality were combined with a view of fire as an enemy — an evil force producing chaos.

I will offer some insights into this curious mixture of scientific knowledge and mysticism as a way of emphasizing the novelty of modern fire management. But equally important, these insights will caution fire managers to avoid embracing new crusades that may prove to be as irrational as the attempt to exclude all fire from wildland ecosystems.

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A Paradox

I have spent over a decade attempting to resolve a paradox. Let me state this paradox in the form of a question: Why would a people so imbued with independence, practicality, and scientific rationality call upon the symbolism of the enemy -- especially the devil-- in an attempt to control wildfire? Until well into the second half of this century wildfire was routinely portrayed as a demonic power. Its only products were chaos, destruction, and lingering fear. Therefore, wildfire control required moral vigilance, and the use of the best available scientific knowledge and technology.

This view is clearly symbolized in fire prevention messages prevalent during the first half of this century. An especially revealing fire prevention poster from the 1920's shows a solitary camper situated in a forest with his tent and car. A bright orange fire demon reaches down from where it is intertwined among the trees to release the flames of the campfire into the forest. The cultural message is clear. A firm rational foundation is symbolized by the rigid trees, car, and tent. Yet the carelessness of a solitary individual has released the powers of a demonic force.

We are reminded of what Smokey will say years later: "Remember, only YOU can prevent forest fires." Was Smokey invoking the moral rigidity of the singular YOU, and calling for the watchfulness earlier symbolized by an individual resisting the temptations of the Devil? Since I am about to get ahead of myself, let me frame the problem a bit more broadly before recalling the imagery of moral danger and individual vigilance we have historically associated with wildfire suppression.

Power of Myth

The history of fire control in the United States has been captured masterfully by Stephen Pyne in a book entitled Fire in American: A Cultural History of Wildland and Rural Fire. Pyne was aware of the same contradiction. He chronicled the development of wildfire control, and likened it to a "military policing action, or a militarylike esprit de corps, or a military organization" (p.238). He noted that the distinguishing characteristic of wildfire control since its inception early in the century has been combative posture appropriate only to a protracted struggle with an enemy.

Pyne attempted to explain the militaristic history of fire control from the viewpoint of a cultural historian. In particular, he described the "climate of opinion" during the emergence of scientifically-based conservation in the Progressive Era, and uses an essay written by the American philosopher William James to illustrate how a spirit of militarism and hardihood prevailed.

James was a pacifist who harbored the painful memory of the Civil War, but admitted that the martial spirit inspired by a common enemy could not and should not be eradicated. Instead of killing one's fellow humans, James advocated a war against nature that would subdue the earth and make it more fruitful in serving human purposes. His essay, "On the Moral Equivalent of War", was celebrated for its insightfulness when it first appeared in 1910, and was rediscovered in the painful years following the war in Viet Nam. Pyne identified in the ideas of James a cultural motif that found repeated expression in the noble and heroic challenge of combating wildfire.

As a cultural historian, Pyne was not troubled by the incongruities involved in intertwining scientific rationality with militaristic spirit. Such complexity in cultural themes makes history all the more interesting. Like a complex tapestry, a complex culture is enlivened by its diversity of contrasting themes.

However, as a social scientist, I have been troubled by an apparent mystical synthesis between demonic symbolism and technical knowledge that has typified wildfire control. The Progressive Era initiated a conservation crusade in which science and technology were tools for combating a demonic power -- enemy fire! Neither Pyne's cultural history, nor James' pragmatic acceptance of the martial spirit, provides a satisfactory explanation for this most unlikely marriage.

After having sought unsuccessfully for explanations in social history and the development of institutions that exclusively embody scientific rationality (Lee, 1976), I have turned increasingly toward depth psychology and the analysis of myth. The study of myth offers a powerful vehicle for understanding paradoxes such as the marriage of technical knowledge and demonic symbolism in fire control.

The meaning of the word "myth" needs to be enlarged. According to the Concise Oxford Dictionary, a myth is "a Purely fictitious narrative usually involving supernatural persons, actions, or events, and embodying some popular idea concerning natural or historical phenomena." This meaning closely resembles its everyday usage. Romulus' slaying of Remus as the event that empowered the founding of Rome, Prometheus stealing fire from the gods, and Santa Claus are all myths. Although Romulus and Remus or Prometheus do not stir our souls, Santa Claus reawakens the childhood mystery of Christmas in very rational adults.

For those who lived through the Great Depression and World War II, the identification of wildfire suppression with control over a demonic enemy still moves the imagination. In 1942 the Wartime Advertising Council joined forces with the U.S. Forest Service to launch a fire prevention campaign that substituted characteristics of the German and Japanese leaders for earlier symbols of the devil. The slogan "Careless Matches Aid the Axis" captioned the leering face of the "foreign devil" with a match standing in front of a wall of flames. The myth of a malevolent fire demon may be "purely fictitious narrative", but it has all the force of a living myth for those whose souls are imprinted by an association of wildfire control with the triumph over hopelessness and despair from the 1930's, exhilirating fear of chaos and destruction from the 1940's, and reawakening of that fear during the Cold War of the 1950's. A resurgence of the cold war in the 1980's reminds us again of the power with which the myth of evil forces inspires collective action in the face of a threat.

We are discovering that the modern age is dominated by living myths. Certain symbols, certain ideas, certain images, have an overpowering effect in mobilizing entire peoples for concerted action. The fire demon served as James' "moral equivalent of war" in eliciting a common response from the mass of Americans at about the same time that a hysterical little man with a comic moustache started talking of blood and soil, of living space, of a mystical unity, Ein Reich, ein Volk, ein Fuhrer, and moved the imagination of a people to such an extent that it altered the whole course of history. Both events illustrate the force of living myths. Social scientists try to neutralize these myths by calling them "ideologies."

I am now moving toward resolution of the paradox of demonic symbolism and technical knowledge in fire control. The act of calling up symbols of destructive, chaotic, and malevolent forces, especially the devil, has provided a moral imperative for directing scientific knowledge and technology. As social scientists and historians, we have been remiss in not accounting for such moral evaluations. We have confined our study to non-moral questions such as "How did society evolve to survive under certain conditions?" or "How can a society be structured so that it will survive under uncertain future conditions?" Study of fire control practices has been restricted by such questions.

We have not attempted to provide an answer to the question "What is the good society?" As a result, we have not adequately understood the role of myth in the collective lives of people in society. Our partial questions yield an impoverished understanding of how people act together to accomplish projects such as fire control, national defense, or extraction of resources from nature. But, most importantly, by ignoring the importance of moral evaluations, we have been unable to understand why reason supplants myth. We have not been able to track the process of moral development. I am now prepared to ask the question that is the title of this talk: Can reason suppress the fire demon?

Moral Development and Fire Control

I will now evaluate wildfire control as a form of human behavior. There are many competing standards for assessing the morality of human behavior. Some standards are derived from religions, and assign the ultimate authority for judging good and bad to the presence of a deity. Other standards are derived from theories of the state, and place authority for judging behavior in a form of government. Other standards are based on theories of individual welfare; the hedonism of modern utilitarian ethics tells us that good is defined by what makes us feel pleasure, so long as we don't hurt others in the process. For purposes of evaluating fire control, I have derived my standards from democratic ethics and scientific rationality, as represented by the thought of Sir Peter Medawar, a British Nobel Lauriat in Medicine.

Medawar (1982) reminds us that the emergence of reason in the Age of Enlightenment was originally based on the conviction that reason was _necessary_ for making human progress. Only later did zealots for rationality argue that reason was _sufficient_, and could replace religion and other sources of knowledge and moral purpose. By emphasizing the necessity rather than the sufficiency of reason, Medawar provides us with the means to assess the relative contributions of scientific rationality _and_ moral principles in social conduct -- in this case wildfire control. Significantly, Medawar reminds us that the ability to reason is essential for exercising democratic governance.

Most importantly, we can examine the possibility that moral progress has been made by substituting a broader use of rationality for a crusade against a destructive and chaotic power. Let us now look more closely at the question of whather reason can suppress the fire demon.

Perhaps we should first learn more about the nature of demons as mythical creatures. According to The _Columbia Encyclopedia_, belief in demons originated in:

> a desire to reduce the incomprehensible, impersonal, often chaotic forces of nature into concrete forms. By endowing these forces with definite shapes and with human emotions, man is better able to enter into communion with them and to propitiate them. The early Egyptians considered storms and the terrors of the night to be the anger of the serpent-demon Apep; not only did this belief explain the origin and meaning of storms, but it also implied that storms could be dispelled if the anger of Apep was sufficiently pacified by sacrifice or ritual. (p. 558)

I would be foolish to assert that the act of evoking a fire-demon is evidence that there has been little moral progress since ancient Egyptian times. Fire suppression is a thoroughly practical task, devoid of the sacrifice of humans and animals, or other rituals designed to appease an angry demon. However, I believe I am safe in asserting that fear of fire as a chaotic and evil power has been captured by the symbol of the demon, and that this symbolism has served a profound political function by mobilizing masses of people in opposition to wildfire as an all-destructive force in nature. Since humans are frequently the source of ignition for wildfires, they are blamed for releasing these chaotic forces, thereby permitting fire to inflict destruction on forests, wildlife, watersheds, improvements, and human life: "Remember: Only _You_ Can Prevent Forest Fires!"

Yet people have grown increasingly aware that the role of humans in starting destructive wildfires is only part of a complex problem. The application of science to fire control has revealed that not all wildfires are started by humans and that not all fires are bad -- some fires may actually be necessary for maintaining natural communities that are valued for their vegetation or the habitat they afford valued wildlife species. Moreover, scientific research has demonstrated adverse consequences of keeping fire out of vegetative communities where it burned at fairly regular intervals under natural conditions. Fire control has allowed an unprecedented accumulation of living and dead vegetation, in which fires burn so intensely as to destroy trees that survived periodic smaller fires for hundreds of years. In addition, the application of economic analysis to fire control has shown that we regularly spend far more to extinguish wildfires than would be lost by permitting them to burn to natural boundaries or practical control lines; spending 500 dollars to save one dollar of resource values has not been unusual.

The replacement of "fire control" by modern "fire management" signifies the dual role that scientific rationality and democratic ethics have come to play in our response to fires in natural environments. We seldom hear talk of "enemy-fire" or the "fire-demons", and even Smokey seems a bit wimpy these days. Far more importantly, we are using our ability to reason to determine what fire can do to help us better serve the multiple purposes for which we manage wildlands in a complex democratic society. But, the question remains: can reason suppress the fire demon?

My answer to this question is: I don't know. It all depends -- it all depends on whether we as a society can find means of social governance that will substitute effectively for mass mobilization in the face of a common enemy. Clearly, we need some form of social coordination. We cannot permit everyone to "do their own thing" with fire in natural environments. Our first step in searching for a substitute for a common enemy will be to subject the dynamics of such mass mobilization to the forces of reason by asking how such moral progress has been possible in the past.

Recent scholarship provides us with important insights into how moral progress became possible in Western civilization when the chains of myth were broken and people began to reason for themselves. According to Maccoby (1982), the Bible can be read as a chronicle of such moral progress. Moral development began in societies in which people's lives were governed by myth and ritual; people's motives --especially good and evil-- were projected upon real or imaginary objects that were thought to control life's events. Ritual was employed to appease demons and seek the favor of beneficial gods -- first involving human sacrifices and then the sacrifice of animals and other objects that were thought to participate in this religious cosmos. Later in the Biblical account of history individuals began to reason and assumed responsibility for the propagation of good and evil. Individuals recognized in themselves properties that had previously been attributed to specialized gods and demons. It was then possible for people to become agents for creating changes in the world by assuming responsibility for the consequences of their actions. This moral progress is the social (as distinguished from the spiritual) essence of the Judao Christian heritage that has had such a profound influence on the Western democratic culture in which we find ourselves. Today, people from other nations, and somewhat surprisingly, even some of our own citizens, fail to understand the origins of the moral strength we derive from self-criticism.

Our broadened capability to reason has recently enabled us to take responsibility for fire as a natural process in wildland ecosystems. We now recognize that whether fire is good or bad depends on what purposes we want to achieve, how much money we have to spend to achieve them, and how much resource value we will lose by permitting a fire to burn. We have progressed beyond the simplistic world where every fire threatened chaos and had to be suppressed at any cost. The chains of myth have stretched to provide us movement. But can the chains be broken so that reason will prevail as an instrument for serving the diverse needs of a democratic society?

Conclusion

Unfortunately, the answer to this question will elude the influence of fire management researchers, resource specialists, and practitioners. The success of fire management will depend on how well the society at large can learn to regulate itself without resorting to attributions of evil as an instrument for maintaining or expanding internal cohesion and social control -- or, stated positively, success will depend on the prospects for democracy as a political institution -- an institution based on the moral principle that individuals are free to pursue their own interests so long as they also take responsibility for the adverse effects of their actions on others. A society in which individuals are self-governing within a framework of laws is a society that does not need the unity provided by a common enemy or demonic force.

This is where I depart from William James and his "Moral Equivalent of War." The martial spirit is not an inevitable, inborn impulse, since it is instead a very human response to fear a threat that is poorly understood and to which malevolent attributes have been assigned. We have learned that there is no reason to respond to wildfire with an overwhelming sense of fear when its causes, behavior, and effects are known scientifically and are, at least partially, controlled by modern technology. Yet we will have to credit William James for his insights if we revert to earlier habits of sublimating in firefighting our mortal combat with "evil" forces in the world at large!

As a scientist, I have felt uncomfortable in asking moral questions about society. However, I have found that human social behavior cannot be fully understood without explicitly taking a moral position from which to ask nonmoral questions. The intertwining of technical knowledge and demonic symbolism remains paradoxical only if the observer insists on moral neutrality. I have resolved the paradox by recognizing the necessary, but insufficient, role that reason plays in human affairs.

Finally, I have argued that democratic governance, with its emphasis on the morality of individual initiative and the necessity for reason, affords uses of reason that are superior to a moral order involving mass mobilization in response to the mysterious workings of malevolent powers.

But before closing, I feel obligated to suggest that another myth is emerging to weaken the tenuous hold that rationality and democratic purpose have on modern fire management. Like granola and jogging, fire is now widely assumed to be "good" because it is "natural." There is a danger that a crusade of "burners" will succeed the fire demon after a brief interlude in which rationality served the social purposes of fire management. Can moral progress be sustained by resisting the appealing simplicity of myth? I really don't know, but I think we ought to try.

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A Multidisciplinary Approach to Evaluating Fire Effects

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Abstract

Federal resource managers are required to take a holistic view of management activities and the results of those activities. Yet, few managers have the funds or manpower required for detailed evaluations of the many disciplines involved. This discussion attempts to justify the use of the plant successional stage as an integrating factor which can be related to conditions in several disciplines. This is done using readily available or easily obtained information. A method for selecting a relatively small set of critically important variables and relationships from among a much larger set of potential variables is outlined. A possible framework for acquiring and organizing data specific to local conditions and management objectives is also suggested.

Introduction

Public land managers are no longer simply managers of grazing or timber resources. They are required to take a more holistic view of management activities and their results. The Federal Land Policy and Management Act of 1976 (U.S. Laws, Statutes, etc., Public Law 90-2743) and the National Forest Management Act of 1976 (U.S. Laws, Statutes, etc., Public Law 94-588) require detailed and holistic plans be prepared for management of public rangelands and forests. The National Environmental Policy Act of 1969 (U.S. Laws, Statutes, etc., Public Law 91-190) requires that the environmental consequences of planned actions involving expenditures of Federal funds be examined and revealed.

One of the weakest aspects of the planning and assessment process has been an inability to predict or efficiently assess the effects of management practices on the many facets of the environment. This has resulted in the criticism of land-use plans and environmental impact statements by members of the public, other agencies, and the courts. Better techniques are needed to enable resource managers to predict and assess the impacts of various management practices on all aspects of the rangeland environment. This is made more difficult by the interactive nature of a tremendous number of variables, but a more systematic method for looking at the variables and interactions may be helpful.

A large amount of literature is available on the effects of fire on various environmental parameters. Yet, it is increasingly difficult for resource managers, with many other duties, to become familiar with the literature relevant to the many aspects of their jobs. Managers need a conceptual framework which will aid them in assessing the impacts of fire on vegetation, soils, wildlife, livestock, aquatic systems, recreational values, or any other concern. Possibly, the greatest problem is to bring together existing information so it can be effectively used by managers. Certainly, much additional research is needed, but resource managers must work with what is presently available.

Fire effects can be evaluated in many ways, ranging from very specific and intensive measurements of various components of the major disciplines to simply relying on past experience to "estimate" the influence of fire on the environment and land-use practices. Both of these methods have their place in land management. However, what is needed is a procedure to help managers determine the critically important relationships and focus their limited resources on understanding those relationships.

Four objectives were used in the development of this fire-effects evaluation framework, designed for use by resource managers. Those objective are:

(1) Avoid simply reviewing specific evaluation techniques. That information is available from other sources.

(2) Find and justify the use of a final integrating factor which can be related to conditions in several other disciplines;

(3) Use information readily available to or easily obtained by resource managers as the integrating factor; and

(4) Suggest a possible framework for resource managers to use for acquiring, organizing, and using data specific to their areas and management objectives.

The Integrating Factor

Fire can cause numerous measurable changes on a sagebrush (Artemisia spp.) site. As a result of fire, parameters such as total vegetation cover, infiltration rate, runoff, sediment load, species diversity or composition (both plant and animal), forage production, livestock production, wildlife density, stream water temperature, and aesthetic appeal may change in one direction or another. All of these parameters, and many others, can be directly measured with many techniques. However, most resource managers have neither the time nor funding for detailed studies. By finding and observing a parameter which reflects the fire-induced changes in many of the

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other parameters, a resource manager can obtain a quick approximation of the overall effect of fire.

Successional stages of plant communities have unique conditions which may result in important differences with respect to wildlife or livestock production, soil stability, watershed conditions, recreational activities, economics, aesthetics, or other concerns. These different conditions are the result of a plant community and its successional stage, as well as soil type, precipitation regime, microclimate, slope, aspect, elevation, and temperature regime. The complex interactions of factors could be broken down and more precise effects of each could be determined. However, even if this information existed for all sites, communities, and successional stages, it would be too complex and unwieldy to be readily used.

Wildlife. Traditional grazing and active fire suppression have greatly influenced sagebrush-grass successional stages. As a result, much of the zone is in poor condition, with sparse understories and heavy dominance of the overstory by sagebrush. Most wildlife species do best where stands have good mixtures of shrubs, forbs, and grasses and/or a variety of cover types in a relatively small area (Urness 1978). Using vegetation type or successional stage to evaluate wildlife habitat is a routine practice in many areas. Yoakum (1980) developed a pronghorn habitat suitability guide based on range condition of sagebrush-grass communities. Several other habitat classification and evaluation procedures have been at least partially based on the relationship between plant successional stage and wildlife response (Whitaker et al. 1976, Thomas 1979, U.S. Department of Interior, Fish and wildlife Service 1980, 1981).

Livestock. Livestock production is strongly influenced by plant successional stage. Removal of tall, thick sagebrush plants by fire will greatly enhance movement of livestock and usually increases grasses and forbs by reducing competition (Wright and Bailey 1982). A good mix of plant species is an important and positive factor determining the quality of livestock diets on rangeland (Rittenhouse and Vavra 1978).

Watershed and Soil. Watershed and soil conditions are greatly influenced by vegetative conditions. Where windspeeds are high vegetation height controls the depth of snow. Thus, the presence or absence of sagebrush plants may have a major influence on overall hydrologic performance of sagebrush lands (Sturgis 1977). Both Hutchinson (1965) and Sturgis (1977) indicated that where drifting was common, the rate of snow accumulation was less for herbaceous vegetation than for sagebrush vegetation.

Osborn et al. (1978) found as much as 10 times greater sediment yields from brushcovered watersheds than from grass-covered watersheds. Soil losses following fire are influenced by rainfall intensity (Orr 1977), size and frequency of bare areas (Packer 1951), soil type, topography, and plant cover (Smith and Wischmeier 1962, Farmer and Van Haveren 1971, Swanson and Buckhouse 1984). Of these factors, vegetation cover and slope are often most important under intense rainfall storms (Farmer and Van Haveren 1971, Meeuwig 1970, 1971). A vegetation cover of 60% to 70% is considered necessary for soil stability (Packer 1963, Orr 1970). Wright et al. (1976) working in west Texas, found that runoff water quality increased and soil loss rates were reduced significantly when plant cover reached 63% to 68%.

Pfister (1981) stated that because of the diversity in the sagebrush ecosystem a finer classification system should be used for improved management of western watersheds. Swanson and Buckhouse (1984) studied several communities of *Artemisia tridentata* ssp. *tridentata*, *A. tridentata* ssp. *wyomingensis*, and *A. tridentata* ssp. *vaseyana* to determine differences in runoff, soil loss, organic and ammonium nitrogen losses. They found big sagebrush communities to be highly variable in soil and vegetation characteristics and in its response to high intensity simulated rainfall. The simulated rainfall caused more soil loss on *A. tridentata* ssp. *wyomingensis* than on the *A. tridentata* ssp. *vaseyana* plots. Reduced runoff and soil loss was partially explained by abundant ground cover.

Riparian and Aquatic Systems. Riparian and aquatic systems are also influenced by the condition or successional stage of adjacent sagebrush-grass rangeland. Low-condition rangeland will have greater peak flows and sediment loads of runoff than higher condition rangeland. However, the riparian vegetation must be given strong consideration when evaluating fire effects of riparian and aquatic systems. Riparian vegetation protects the streambank in three ways: (1) by reducing water speed, (2) by protecting streambank from water borne objects, and (3) by inducing deposition of sediment along the bank (Parsons 1963). Cover is one of the most important habitat requirements of fish. Binns (1978) found cover to be significant ($P<0.01$) for determining fish biomass in Wyoming streams. Boussu (1954) increased trout biomass 258% by simulating riparian vegetation cover over a stream. He also found that destruction of undercut banks reduced trout biomass 66%.

Economics. Rangeland fire management programs are increasingly being judged by their economic efficiency (Bratten 1982). A multidisciplinary team at the Pacific Southwest Forest and Range Experiment station in Riverside, California is developing a non-site-specific analytical and planning approach called the Fire Economics Evaluation System (FEEs)(Mills and Bratten 1982). This approach uses probability models for climate zones, vegetation class, and many other factors (Bratten 1982) and considers the relative management importance of various land resources in the area of concern. These models were designed to be a part of the planning process evaluating potential fire management plans. However, they may be helpful in determining economic impacts following wildfires. Economic considerations of prescribed burning have also been discussed by Workman (1976) and Nielsen (1979). The composition and condition of the resulting vegetation is a major concern in virtually all economic analyses of prescribed fires or wildfires.

Successional Stages

Plant successional stage and condition can be considered as the best readily available "final integrators" with which to evaluate potential uses and productivity of the post-burn site. Using successional stage and condition as the final integrating factors has several advantages. This allows specialists in the various disciplines to translate standard vegetation inventories into information on their specific discipline as long as they recognize that differences can occur within a single successional stage and condition class. Specialists in different disciplines can more easily communicate with each other because they are working and making decisions based on the same data. Much of the information in the various disciplines is already based, to some extent, on plant successional stage and condition. However, a more unified approach to handling and interpreting this information might be helpful.

Blaisdell et al. (1982) outlined four condition classes for big sagebrush ranges. These easily recognized condition classes are used as successional stage descriptions in the present discussion. However, other successional stage schemes or even community types may be used if the categories have sufficient differences with respect to the various disciplines of concern.

Successional Stage 1 is sagebrush with a good understory of perennial grasses and forbs. These ranges have not changed greatly from their original condition, and forage production is not far below the potential. Palatable perennial grasses and forbs make up the understory and comprise more than a third of the total vegetation. Soils are essentially unchanged from the original condition with no observable erosion. Condition is classed as good or excellent (Blaisdell et al. 1982).

Successional Stage 2 is sagebrush with a sparse understory of perennial grasses. Sagebrush is greatly increased, perennial grasses are reduced to scattered stands, and perennial forbs are lacking. Erosion is often severe on sloped sites and may be unchanged on level sites. Forage production is light and mostly unavailable to grazing animals. Range condition is poor to fair (Blaisdell et al. 1982).

Successional Stage 3 is typified by sagebrush with an understory of annual grasses and forbs. These ranges have dense stands of sagebrush with only a few scattered perennial herbaceous plants. Severe erosion has often occurred on these sites. Pedestalled plants and erosion pavement may be common. Forage production is poor and fluctuates greatly between years. Range condition is considered to be poor.

Successional Stage 4 is recognized as rangeland with sagebrush replaced by cheatgrass or other annuals. Overgrazing, recurrent wildfires, or cultivation have resulted in the loss of sagebrush. Soil loss may be severe on some sites. Forage production is very seasonal and may be extremely low in years with low precipitation. Range condition is very poor.

Predicting Successional Stages. Planning a prescribed fire requires resource managers be able to select the proper time, conditions, and procedures for burning in order to obtain the desired results. Prescribing a fire to accomplish the desired selectivity requires careful consideration of many factors and is beyond the scope of this discussion. However, this information is available from several research reports and reviews (Blaisdell 1953, Blaisdell and Mueggler 1956, Mueggler 1956, Conrad and Poulton 1966, Wright and Klemmedson 1965, Johnson and Payne 1968, Wright 1971, Harniss and Murray 1973, Uresk et al. 1976, Wright and Bailey 1982) in addition to many of the other papers in this symposium.

Organizing Information

Efficiently using the large amount of information concerning fire effects necessitates development of a useful framework for organizing facts and presenting them in a lucid manner. This requires placing the information into a readily available and easy-to-use format. Another problem faced in predicting or assessing the consequences of a prescribed fire or a wildfire is to know that all the major points have been considered.

One approach to displaying information which gives a quick overview of the entire situation and aids in consideration of all interactions is the matrix approach described by Luna Leopold et al. (1971). The matrix approach to information handling has been used for environmental impact assessments and has been successfully used by the U.S. Forest Service as a wildlife habitat decision-making tool in the Blue Mountains of Oregon and Washington (Thomas 1979). The use of a matrix as an information display can be an effective way to summarize large amounts of information into a relatively easy-to-use format. However, a certain amount of information is inevitably lost with summarization.

Several approaches involving the use of matrices have been proposed in agency guidelines and by various research groups. The matrix might be visualized as a two-dimensional checklist and is the first step toward systematically defining interrelationships (Leopold 1974). Construction of a well-designed matrix assists in systematically evaluating fire effects.

A study of fire effects must begin by defining a set of variables and associated relationships of importance in understanding the problem of concern. Even describing a relatively small system, in detail, could easily involve hundreds of variables and thousands of interactions. It is neither feasible, practical, or even useful to fully evaluate all the possible interactions. Accumulating data on individual variables is less important than focusing on data that can be used to understand relationships.

Bonnicksen and Becker (1983) developed a procedure to select a relatively small set of critically important variables and relationships from among a much larger set of potential variables. This procedure is also used to assign priorities to variables that deal with the relationships described by the system. This system is referred to as the cross-impact assessment process (CAP) and was designed for use in any assessment problem that can be defined by a set of variables and relationships. CAP may provide an efficient and reasonably objective means for assigning priorities to studies designed to assess fire effects on rangelands.

CAP is based on a series of structured workshops. The workshop participants ideally represent a variety of disciplines and are selected because they have both the depth of understanding necessary to contribute specific knowledge and the breadth of view to integrate that knowledge with information provided by other panalists (Bonnicksen and Becker 1983). The first two steps in CAP involve identifying relevant variables and hypothesizing the relationships that tie these variables together (Kane et al. 1973). Panels are used to identify a small set of critically important variables affecting all disciplines of concern and to hypothesize relationships. Discussing the details of the CAP process is beyond the scope of this paper, but the process is described by Bonnicksen and Becker (1983).

The formal CAP process might be useful for large projects. However, most of the smaller assessment jobs confronting resource managers could be adequately handled with a similar, but less formal, meeting of available specialists from several disciplines.

Matrices should be constructed with successional stages or condition classes listed across the horizontal axis and disciplines or discipline components down the vertical axis. Tables 1 and 2 are examples of these types of matrices. The parameters in the vertical axis of the matrices should have a definable relationship with the different successional stages listed across the horizontal axis. A short status statement is made in each matrix cell for each combination of successional stage and discipline component.

By arranging this information into differing levels of specificity, the resource manager can acquire an overview at one level, a more detailed summary at another level, and be directed to several references at yet another level.

Hierarchical Arrangement of Information

Depending on their specific requirements, resource managers can obtain information at four levels of detail (Figure 1). The amount of detail increases with each level: Level 1 - A qualitative expression of the general effects of fire on relationships between successional stage and each of the major disciplines of concern (i.e. soils, watershed, wildlife, livestock, etc.). Level 1 might consist of a single sheet of paper (Table 1) with successional stages on the horizontal axis and major disciplines of concern down the vertical axis. The summary of overall effects can be displayed for each cell of the matrix. Level 2 - Relationships of plant successional stages to various components of the major disciplines. Level 2 might consist of one page for each discipline with successional stages listed on one axis and the various components of the selected discipline of the other axis (Table 2). Level 3 - Summary of relevant data for a selected discipline which may be several pages in length. Level 4 - Selected references for each component of the selected discipline. This level contains the literature which was cited in level 3. This may consist of published research or notes from personnel working in the area of concern. Resource managers can use this information to find published reviews or detailed information on specific problems.

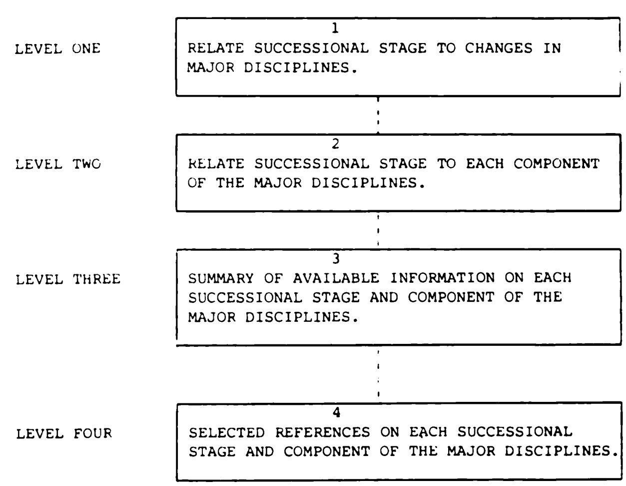

Fig. 1. Hierarchical arrangement of information relating fire effects to components of several disciplines for each successional stage.

Table 1. Example of level-one matrix for big sagebrush range.

Discipline	Successional stage: Sagebrush with good understory of perennial grasses and forbs.	Sagebrush with sparse understory of perennial grasses and forbs.	Dense sagebrush with understory of annual plants.	Dominated by annual grasses. Little or no sagebrush present.
Soils and watersheds	Relatively unchanged from pristine conditions.	Few problems on level sites, erosion may be severe on steeper sites.	Greatly increased potential for erosion.	Possibility of severe erosion.
Wildlife	Relatively good for most species.	Reduced potential for most species. May provide good habitat for some species.	Poor habitat for most species. May provide good winter range for pronghorn and sage grouse.	Low quality habitat for most species. May provide seasonal forage for herbivores and seeds for grainivores.
Riparian and aquatic systems	Good condition providing streambank vegetation is not damaged.	Varies with topography. May have increased sediment load.	Usually has increased sediment load. As riparian vegetation is lost, water temperature bank cutting, and water water speed increases.	Often accompanied by loss of much riparian vegetation. Water quality and depth increase. Poor habitat for native fish.
Livestock	Good for all kinds of domestic livestock Diet quality increases with plant species diversity.	Reduced livestock performance resulting from loss of perennial grasses and forbs.	Low quality livestock range.	Only of very seasonal benefit.

Table 2. Simplified example of level-two matrix relating sagebrush successional stage to some soil and watershed components.

Soil and watershed components	Successional stage: Sagebrush with good understory of perennial grasses and forbs.	Sagebrush with sparse understory of perennial grasses and forbs.	Dense sagebrush with understory of annual plants.	Dominated by annual grasses. Little or no sagebrush present.
Shrub cover	10-30%. Several species may be present.	20-40%. Mostly sagebrush.	30-60%. Almost exclusively sagebrush. May have extensive areas of rabbitbrush.	Little or no shrub cover.
Herbaceous plant cover	Good representation of perennial grasses and forbs.	Poor representation of perennial, herbaceous plants.	Few perennial, herbaceous plants. Many annuals.	Almost exclusively annual plants.
Infiltration	At or near potential.	Moderate to low. Varies with soil type and litter cover.	High interception losses because of shrub density. Low infiltration between shrubs due to lack of herbaceous plants.	Poor, except in coppice mounds of dead shrubs.
Runoff potential	Varies with topography. Usually relatively low.	Moderate to high.	High on sloped sites.	High.
Erosion potential	Little or none on most sites.	May be severe on steeply sloped sites and unchanged on level sites.	Pedestalled plants and erosion pavement common.	May have severe soil loss.

These information matrices, summary sheets, and specific literature can be constructed from: (1) the scientific literature, (2) interpretation and extrapolation of information in the literature, and (3) the consensus of people working in the discipline. In each case, the best information should be used. Previously developed evaluation procedures, such as one of the wildlife habitat evaluation schemes, can easily be incorporated into this framework. When site-specific information is available, it should be included. Seldom will such site-specific information be available for all areas of interest. Appropriate information from other areas will have to comprise much of the information. In any case, the information must be considered as a beginning which will be under constant revision and update.

The specific information needed for this approach cannot be produced for a large area because it is fairly specific for local sites, objectives, and communities. As a result, considerable local effort for the preparation of a usable product is required. However, it forces knowledgeable and experienced personnel to record information in a way that it can be easily used by others. It encourages specialists to interact and discuss relationships and interactions.

This evaluation scheme evaluates range sites by successional stage. It makes no difference whether that stage is a result of fire, grazing management, or some other influence. Fire can cause unique short-term responses which should be considered and incorporated into the information base. Evaluation by successional stage is designed for general planning and evaluation purposes and does not automatically consider specific problems. Situations involving post-fire responses of endangered species, unique or critical areas, litigation, or some other unique facet of the resource may not be adequately handled with this approach alone. These situations may require a much more detailed analysis. However, the information gathered for these information displays may aid in designing a more detailed study. This detailed information, or at least a mention of the potential problem, should be incorporated into the information matrix when available.

The resulting information comprises a data base from which resource managers may obtain information at various levels. A similar system, designed for wildlife habitat management in commercial forests has been well tested and has enabled users to produce better, more comprehensive, and more accurate reports in less time (Thomas 1979).

This method of organizing a specific data base cannot replace trained and experienced specialists in each of the relevant disciplines. This unified approach to evaluation is designed for general planning and evaluation purposes. Its advantages are speed, economy, and flexibility. The principle disadvantage is the lack of precision when applied to very specific problems. It is a tool for the resource manager, which if applied without proper interpretation to the specific conditions will not be as accurate as possible.

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Improved Low Intensity Soil Survey Map Unit Design Using Large Scale Vertical Aerial Photographs

Karl W. Hipple and W. D. Harrison

Many field soil scientists have labored through the map unit design process only to have the correlator question the identified components or even change the soil map unit during the field review process. Transecting soil map units for documentation is challenging, trying, and frustrating, especially in remote, inaccessible areas. A useful "tool" to supplement map unit design and documentation procedures is available - large-scale aerial photography. With these photographs, the soil scientist can identify, document, and quantify soil components where soil differences can be related to vegetation, landforms, or topographic feature indicators.

Field sheets for most Idaho soil surveys are 1:24,000 scale "third generation" aerial photographs. Landscape features or patterns can be recognized during the premapping process, but often photo scale restricts actual quantification of the identified features. In most cases, these areas are quantified, and supportive documentation is gathered after the soil scientist goes to the field. Only then can percent map unit composition (major and minor components plus inclusions) be identified and transects, traverses, and observations completed to document the soil map unit.

Map unit notes generally take the form of soil observation notes, landscape diagrams, and soil and landscape photographs taken from the most advantageous point; such as, a prominent landscape feature, the ground, or a pickup cab.

A method developed by Meyer and Grumstrup (1978), can be used to obtain large scale vertical 35 mm aerial photographs to study soil, vegetative and miscellaneous land type patterns that are not visible, or are visible but not quantifiable, on standard 1:24,000 soil survey field sheets. The large scale vertical 35 mm aerial photographs are relatively low cost and aid soil map unit design by providing additional documentation for soil map units. The method offers short waiting times from the actual photography to a finished usable product (usually 10 to 14 days).

The method uses a 35 mm camera system and a portable camera mount attached to helicopters or "high-wing" single engine aircraft. The flexibility of this method offers many options for scale, end-lap, and films. Photo scale is variable by changing lens focal length or platform height above the ground. Stereoscopic end-lap is variable in the flight line by changing aircraft speed and/or exposure interval. Film variety and filter combinations are also available to obtain desired effects. The imagery is typically flown in single line transects. Transect length is determined by the number of exposures per film roll and/or the desired amount of stereoscopic end-lap. These decisions are approximated during premapping activities.

Currently, this method is being tested in Idaho in the Custer-Lemhi Soil Survey Area. This area contains large acreages of fan terrace landforms which exhibit varying amounts of mound-intermound features. These fan terraces have been dissected to varying degrees in response to nearby mountain uplift and runoff during pleistocene glacial events. The areas adjacent to the dissections exhibit varying amounts of windswept vegetation features. Although these areas can be visually identified on 1:24,000 scale soil survey field sheets, the mound-intermound patterns and the windswept features are so complex and small in size that quantification of these features is impossible without actual on-the-ground measurement.

The previously described method was used in the Custer-Lemhi area after field mapping was approximately 80 percent completed in an attempt to test the photographic technique and map unit design with respect to percent composition of mound, intermound, and windswept components. Other features which may lend themselves to quantification by this method are miscellaneous land types, such as rock outcrop and rubbleland, and surface features, such as surface stoniness and slickspots.

A Bell B-2 helicopter with side baskets was used. A motor drive Nikon F2 Photomic 35 mm camera equipped with a f/2.8 135 mm lens, a UV-2A filter, and a remote shutter release were attached to the side basket of the helicopter; Kodachrome 64 transparency (slide) film was used for all transects. Transects were flown at 500 and 250 feet above ground level at 60 mph ($\pm$10 mph). This resulted in film scales of 1:1,200 and 1:600, respectively, with 40 to 60 percent stereoscopic end-lap.

Map unit component features were observable and quantifiable from the slides. Photo enlargements were made from the slides and component features were transposed to mylar sheets and measured using a dot grid. Ground transect and photo enlargement measurements were compared for soil and vegetation percent map unit composition.

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In the case of the six transects flown, map unit composition varied by a maximum of 10 percent from the on-ground transects established by the soil survey crew. In five of the six transects the map unit composition varied by only 2 or 3 percent. The 10 percent variability occurred because the location selected for the transect was low on the fan terrace where the dissections were more numerous and pronounced, and the accompanying windswept component was greater. The overall variability between the flown transects and the on-ground transects appears to be within ± 2 to 5 percent.

There are many advantages to using large scale vertical aerial photographs for soil map unit design: 1) Soil map units in remote areas can be more thoroughly observed; 2) Individual bias in map unit design may be reduced; 3) Photographs are permanent records of map unit transects; 4) Time spent completing area reconnaissance may be reduced by studying carefully selected photo-transects; and 5) Extent and component percentages of some map units can be accurately determined as a check to ground transects.

This method does, however, have some limitations. These include: 1) aircraft rental is expensive, so careful planning is necessary; 2) no attempt to infer map unit components can be done without adequate ground verification (this method is not a substitute for established ground documentation procedures); and 3) soil features and vegetative patterns must be known before the interpretation of large scale vertical aerial photographs for predicting soil taxa is possible.

This method has been in use for several years for making rangeland, riparian, and erosion studies. However, its application to low intensity soil surveys is new and unique. It has proven very useful in the Custer-Lemhi Soil Survey Area to help solve a unique set of map unit design problems. Currently, it is being tested in two other soil survey areas in Idaho, one a high intensity (2nd Order) soil survey. Preliminary results show this method will improve map unit design and provide additional quality control in survey areas of medium to high intensity.

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The following is a selected bibliography relative to small format aerial photography. It is provided as additional reference for those interested in learning more about small format aerial photography in natural resource applications:

Small Format Aerial Photography

William H. Anderson and Frank X. Wallner

Introduction

There are many examples of the usefulness of aerial photographs in natural resource investigations. Unfortunately, suitable aerial photographs are not always available, and contracting for new aerial photographs to be flown by a commercial aerial survey firm can be expensive, especially for relatively small areas such as a single farm, a 100-acre wetland tract, or a city park. Do-it-yourself aerial photography using small format cameras (typically 35mm) has been shown to be a viable alternative in many instances, providing valuable aerial photographs on a cost-effective basis.

This bibliography has been prepared to direct the reader to sources of information about successful applications of small format aerial photographs. It is by no means exhaustive, but serves to illustrate typical methods, materials, and analysis procedures in a number of application areas.

To facilitate access to items of limited distribution, the EROS Data Center Library call number has been included as part of the citation. Other items are found in technical journals of wide distribution.

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Woodcock, W. E. 1976. Aerial reconnaissance and photogrammetry with small cameras: Photogrammetric Engineering and Remote Sensing, v. 42, no. 4, p. 503-511. Description of 35 and 70 mm cameras, film, aircraft, and camera mounts that can be used to acquire inexpensive reconnaissance aerial photography.

Zsilinsky, V. G. 1968. Supplemental aerial photography with miniature cameras: text of paper presented to Commission 4 at the 11th Congress of the International Society of Photogrammetry, Lausanne, Switzerland, 14 p. (EDC call number 264.Z8s). An excellent guide to the technical aspects of taking, printing, and viewing of vertical 35 mm aerial photographs.

New Grasses for Rangeland Improvement

W. H. Horton, K. H. Asay, and F. B. Gomm

Abstract

Recently developed cultivars and interspecific hybrids of grasses were tested for adaptation, forage and seed yield, forage quality, and response to grazing. 'Bozoisky-Select' Russian wildrye (Psathyrostachys juncea), a Russian introduction, had better seedling vigor and produced more foliage and seed than Vinall Russian Wildrye.

The crested wheatgrass hybrid, 'Hycrest' (Agropyron cristatum X A. desertorum) established good stands on dry sandy and desert shrub sites. Its seedlings developed more rapidly and were larger than those of either parent species.

Bozoisky-Select and Hycrest are recommended for seeding dry, arid rangelands.

Crested Wheatgrass

Crested wheatgrass is well known for its productivity early in the grazing season, although the forage value rapidly declines in the mid to late summer months. It should be used during the spring and early summer months in a seasonal rotation grazing program with grasses which mature later and remain palatable longer. As a bunchgrass it is an excellent seed producer and is characterized as one of the easiest grasses to establish on semiarid rangeland sites. It is tolerant to cold and drought and moderate amounts of salinity (Asay and Knowles In Press, Rogler 1973). Where adapted crested wheatgrass is very long lived. It has remained productive in many intermountain sites for over 35 years on sites in Utah and Southern Idaho.

Russian Wildrye

Although this species is difficult to establish, it is one of the best sources of rangeland grazing in the Intermountain West and Northern Great Plains. It was introduced into the United States in 1927; however, its value in reseeding rangelands was not fully recognized until the 1950's (Hanson 1972). It is a long lived bunchgrass with abundant basal leaves. Leaves remain green throughout the season dependent on soil moisture. It is very resistant to cold and even more tolerant to drought than many cultivars of crested wheatgrass (Currie and White 1982).

Russian wildrye has not reached its full potential on western rangelands, because of several inherant deficiencies.

1) It is difficult to establish. Seedlings often fail to emerge especially when planted too deep or when the seedbed is poorly prepared.

2) Progress in breeding is limited by the relatively small amount of genetic diversity available to plant breeders working with the species.

3) Seed of Russian wildrye matures early in July and shatters soon after maturity creating seed harvest problems for commercial seed producers.

Testing of recently developed cultivars and interspecific hybrids of grasses from the USDA-ARS Forage and Range Group is sufficiently complete to recommend two new grasses for release.

Bozoisky-Select, Psathyrostachys juncea (Elymus junceus), was developed from plant materials introduced from Russia in 1977. It is taller, more robust, and produces more foliage and seed than Vinall. The seed of Bozoisky-Select is also larger than that of Vinall. It appears to have good seedling vigor and has been readily established on dryland sites throughout the Intermountain area. As long as soil moisture is adequate to promote growth, the leaf forage is highly palatable and nutritious.

The crested wheatgrass cultivar, Hycrest, is a cross between induced tetraploid Agropyron cristatum and natural tetraploid A. desertorum. Hycrest is very robust with large fan shaped seed heads. Its seeds are larger than those of Nordan and it appears to have excellent seedling vigor. Seedlings develop more rapidly and are larger than those of either parent species. The plant is not as leafy and tends to be more coarse than A. cristatum. It has established excellent stands on dry sandy sites and on shadscale-desert shrub sites adjacent to the Great Salt Lake in precipitation areas of 12-25 cm (5-10 in) where it was established in competition with cheatgrass (Bromus tectorum) and halogeton (Halogeton glomerata). Hycrest eliminated many annual weedy plants by fall of the first year of establishment and virtually all during the second year of growth, whereas Nordan and Fairway were still heavily infested with weedy species at the end of the establishment year. However, this advantage of Hycrest over Nordan and Fairway decreased during the second year's growth. Hycrest emerges more rapidly after germination, produces considerably more forage and seed during the year of establishment, and appears to establish better under a wider spectrum of conditions than other crested wheatgrasses.

Materials and Methods

Studies were located at 45 sites from Buffalo, Wyoming, and Challis, Idaho; south to Escalante, Utah. Sites ranged in elevation from more than 2,300 m (7,500 ft.) on the Idaho-Montana border to less than 1,300 m (4,300 ft.) near the shore of the Great Salt Lake on range habitat-type sites representative of mountain brush, sagebrush, juniper, salt desert shrub, mountain meadow, and irrigated pasture. All selections however, were not planted at all sites. Additionally, seeds of the new species have been sent to cooperators throughout the western states, Mexico, and Canada.

Plantings were made from 1978 through 1983. The performance of the new plant materials were compared to varieties and species generally accepted for planting on arid rangelands. Vigor was estimated from physical appearance of plants. Adaptation was determined by estimates of percent stand establishment and survival. Forage production was determined from meter square samples at oven dried weight. Forage samples were ground and analyzed for crude protein as an expression of nutritive quality. Palatability and response to grazing were determined by allowing cattle to graze free choice.

At most sites, plantings were established from transplants and drilling of seed. Transplants were spaced approximately 3 feet apart between rows and 2 feet within rows. Transplants were cultivated between plants within rows to give each plant the most optimum soil and growing conditions available under the prevailing climate and site location. In drilled studies, seeds were planted through a John Deere Flexiplanter equipped with 1-inch depthbands in rows 30 cm apart with approximately 25 live seeds per 30 cm of row. Extra weight added to the drive wheel insured that the soil was firmly packed around the seed. To be successful, the drilled seed had to germinate, establish, and survive against invasion of natural plant competition.

Results and Discussion

Vigor. Based on establishment, amount of growth, and general appearance, the transplants were subjectively rated on a scale of 1 to 100. Bozoisky-Select consistently rated higher than Vinall, and the Hycrest generally ranked higher than its parental types (Table 1).

Table 1. Relative vigor rating of transplants

Species	Ecosystem: Sagebrush (13)[1]	Juniper (4)	Shadscale (3)	Grease-wood (1)	Indian ricegrass (1)
	Vigor (%)[2]				
Nordan	90	86	90	75	75
Fairwav	87	97	82	75	75
Hycrest	97	98	95	85	75
SE	5.1	6.7	6.6	5.8	0
Vinall	88	82	80	75	75
Bozoisky-Select	93	88	92	97	85
SE	3.5	4.2	8.5	15.6	7.1

1 Numbers in parentheses are the number of sites averaged.

2 Ratings were made from 0 to 100, 0-least and 100-most vigorous.

Establishment. Good to excellent stands of all crested wheatgrasses and Russian wildrye were established from seed at most sites (Table 2). Hycrest consistently ranked higher than its parent types, but there was no difference between the two wildryes.

Table 2. Relative ratings of stand establishment at five vegetation ecosystems.

Species	Ecosystem: Sagebrush (13)[1]	Juniper (4)	Shadscale (3)	Grease-wood (1)	Indian ricegrass (1)
	Stand (%)[2]				
Nordan	91	96	91	85	77
Fairway	90	97	89	45	75
Hycrest	94	100	97	88	85
SE	2.1	2.1	4.2	24.0	5.3
Vinall	81	85	78	80	50
Bozoisky-Select	84	86	84	83	60
SE	2.1	0.7	4.2	2.1	7.1

1 Numbers in parentheses are the number of sites averaged.

2 Ratings were made from 0 to 100, 0-least and 100-most vigorous.

Forage production. During the year of establishment, forage yield of Hycrest was 50% higher than the parent wheatgrasses on identical sites while that of Bozoisky-Select was 15% higher than Vinall (Table 3).

Table 3. Average herbage yields of grasses from several range sites.

Species	Average yields from all sites	Average yields at the same sites
	kg/ha	
Nordan	2473 (36)[1]	2326 (4)
Fairway	1934 (26)	2451 (4)
Hycrest	2704 (8)	3651 (4)
SE	395	732
Vinall	1041 (28)	1200 (20)
Bozoisky-Select	1375 (20)	1375 (20)
SE	236	124

1 Numbers in parentheses are the number of samples averaged.

Palatability. The percentage of herbage utilized by cattle was highest for the Russian wildryes and least for the crested wheatgrasses and intermediate wheatgrass (Table 4). At one site where drilled rows and transplants were compared, cattle utilized more of the plants from drilled stands than from transplants.

This observation is not surprising because transplants have the appearance of "wolf-plants," which under normal grazing systems would be utilized less until after the more palatable forage is removed by grazing.

Table 4. Utilization of grasses grazed free-choice by cattle at different vegetation sites.[1]

	Utilization (palatability) rating			
	Juniper-	Juniper-	Sagebrush	
Species	Oak	Sage	Drilled	Transplant
	----------------Percent-----------------			
Nordan	50	70	30	20
Fairway	45	80	35	25
Hycrest	-	-	-	25
Vinall	85	90	90	65
Bozoisky-Select	-	-	85	65

[1] Utilization ratings were made from 0 to 100. 0 = least and 100 most utilization.

Seed weight. Seed of Hycrest crested wheatgrass was heavier than that of the parent species (Table 5). For comparative purposes, it is the same size (by weight) as the induced tetraploid of A. cristatum. The induced tetraploid was used to cross with A. desertorum to form Hycrest.

Table 5. Relative seed weights of several grasses.[1]

Cultivar	Seeds/kg
Nordan	434,000
Fairway	624,000
Hycrest	335,000
Agcr tetraploid	335,000
SE	136,000
Vinall	333,000
Bozoisky	295,000
SE	27,000

[1] Seed was from transplants spaced on 1 meter centers grown near Logan, Utah.

Seed of Bozoisky-Select Russian wildrye is approximately 13% heavier than Vinall, based on comparative seed weights.

Seed yield. Although yields varied greatly from site to site, Bozoisky-Select consistently produced more seed than Vinall (Table 6). Also, Hycrest yielded more seed by weight than either parent species.

Table 6. Estimated seed yield from spaced plants of several grasses.

Species	Yield (kg/ha)
Nordan	250
Fairway	250
Hycrest	300
RS-1	250
RS-2	250
Vinall	200
Bozoisky	275

[1] Plants were on 1 meter centers with approximately 10,000 plants/ha.

Fire Management Potential. Drilled plot plantings were made on ranges where large cheatgrass (Bromus tectorum) areas had burned during mid-summer months of 1983. The research area was prepared during the late fall green-up period of the cheatgrass. Three methods were used for site Preparation--rototilling, scalping, and chemical treatments--after which all areas were seeded with a drill.

Early results show that application of one pint per acre of Roundup[a] immediately prior to seeding on undisturbed seedbeds produced outstanding results. Using this treatment, Hycrest and Nordan crested wheatgrass and Bozoisky-Select and Vinall Russian wildrye had 70-90% stand establishment ratings, compared to 20-40% establishment ratings on other treatments.

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[a] Mention of a trademark name or proprietary product does not constitute endorsement by the USDA and does not imply its approval to the exclusion of other products that may also be suitable.

Factors Influencing Establishment of Seeded Broadleaf Herbs and Shrubs Following Fire

Stephen B. Monsen and E. Durant McArthur

Abstract

Sagebrush, pinyon-juniper, and mountainbrush rangelands occupy extensive areas in the Intermountain region. Climatic conditions vary within these communities, and not all areas are conducive to revegetation. Native herbs and shrubs are often seeded to improve herbage yields and wildlife habitat, especially following fire. Planting success can be improved by utilizing selected seed sources, controlling competition, developing a suitable seedbed, and planting in the proper season. Seed germination features of most broadleaf herbs and shrubs differ from commonly seeded grasses. Thus, grass seeding practices are not always effective in establishing broadleaf herbs and shrubs.

Introduction

Big sagebrush, pinyon-juniper, and mountainbrush lands can be difficult to revegetate because of low erratic patterns of moisture and fluctuating temperatures during seedling establishment. Average annual moisture is frequently used to determine seeding potential (Trewartha 1957). Areas receiving less than 10 to 12 inches of average annual moisture normally are not recommended for planting (Bleak et al. 1965). Since many shrublands receive about this amount of moisture, it is difficult to predict seeding success, particularly when native broadleaf herbs and shrubs are planted (Plummer et al. 1968).

The seedlings of many native herbs and shrubs are less competitive than commonly seeded grasses and introduced weeds. Although the native species may occur on undisturbed rangelands, once removed they are difficult to reestablish. Factors including climate (Jordan 1983), seed germination, and seedbed conditions (Mayer and Poljakoff-Mayber 1982) influence seedling survival. Some pertinent influences will be discussed in this paper.

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Site Condition Comparisons

The sagebrush, pinyon-juniper, and mountainbrush wildlands of the Intermountain region encompass a large geographic area (Kuchler 1964, Bailey 1978). The sagebrush (subgenus _Tridentatae_ of _Artemisia_) types are more extensive than mountainbrush vegetation types, but both are important. Kuchler's (1964) three large sagebrush types (Great Basin sagebrush with three principal subtypes, sagebrush steppe with two subtypes, and wheatgrass-needlegrass shrub-steppe) extend from Washington and Montana to southern California and New Mexico. The pinyon-juniper and mountainbrush areas are smaller in size although the mountainbrush types are usually more diverse. Mountainbrush vegetation types usually occur on uplands associated with sagebrush. The various mountainbrush vegetation types includes, in various combinations, _Cercocarpus_ spp., scrub oak _Quercus_ spp), bigtooth maple (_Acer grandidentatum_), antelope bitterbrush (_Purshia tridentata_), Stansbury cliffrose (_Cowania stansburiana_), mountain big sagebrush (_Artemisia tridentata_ ssp. _vaseyana_), myrtle pachistima (_Pachistima myrsinites_), mallow ninebark (_Physocarpus malvaceus_), Ceanothus (_Ceanothus_ spp.), serviceberry (_Amelanchier_ spp.), common chokecherry (_Prunus virginiana_), bitter cherry (_P. emarginata_), snowberry (_Symphoricarpos_ spp.), and other species.

Shrubs reach dominant status on large tracts only in the arid and semiarid western parts of the United States (Kuchler 1964, McArthur 1984). Stebbins (1972, 1975) listed drought or aridity, nutrient-poor soils, fire, shade, poor soil aeration, winter cold, short growing seasons, and wind as stress conditions that promote shrub dominance.

Although western shrub tracts may be very large considerable species diversity exists within them (McArthur 1984). For example, Hironaka et al. (1983) described 38 separate habitat types for the sagebrush and mountainbrush lands for southern Idaho. Blaisdell et al. (1982) listed 43 habitat types for sagebrush and commented that their list was incomplete. Individual shrub, forb, and grass species of the habitat types or vegetative communities and, consequently, the communities have different fire response behavior (Wright et al. 1979, Table 1).

Table 1. Proportionate fire damage to three plant classes in sagebrush-mountainbrush wildlands.

Life form	Species	Percent damage by fire: Slight or none	Moderate	Severe
	(n)			
Grasses	24	75.0	12.5	12.5
Forbs	42	16.7	35.7	47.6
Shrubs	26	73.1	11.5	15.4

Because of the differential responses of individual species, native vegetation may not respond favorably after a fire. A land manager is faced with a complicated problem when seeding species with different life forms on a burned area.

Distribution of plant species and communities is controlled by climatic and soil conditions (Kuchler 1964, Bailey 1978). These relationships are easily seen on a large scale. However, on small tracts in broken topography, which is common in the Intermountain Region, the relationships are difficult to observe. Beetle and Young (1965), Winward (1980), and McArthur (1983) have described four major subspecies within the big sagebrush complex (Artemisia tridentata ssp. tridentata, [basin], ssp. vaseyana [mountain], spiciformis [subalpine], and wyomingensis [Wyoming]). The different subspecies occur in discrete units, but they also may change gradually so that boundaries are difficult to determine. Pinyon-juniper and mountainbrush communities are also discrete in places, but intergrade with one another and with big sagebrush in other locations (Kuchler 1964; Holmgren 1972).

Table 2 is a comparison of several climatic and topographic factors for three sagebrush communities, and for select pinyon-juniper and mountainbrush communities in Utah. The fourth big sagebrush community (subalpine) was not included because areas it occupies are fewer, smaller, and not of much concern in fire rehabilitation efforts. Weaver (1980) made a comparison similar to Table 2 data for the Northern Rocky Mountains and adjacent plains. Our results and Weaver's (1980) show that some physical factors are unquestionably different in the distinct plant communities, other factors show predictable trends, and no patterns are apparent in others. Factors of note include: 1) basin big sagebrush sites have a longer frost-free period than the others, 2) mountain big sagebrush, pinyon-juniper, and mountainbrush sites are usually at higher elevations; 3) mountain big sagebrush sites generally are colder by all criteria examined; 4) Wyoming big sagebrush sites are the driest (they have less precipitation, more wind, more solar radiation, lower relative humidity, and high evaporation); 5) mountainbrush and mountain big sagebrush sites have higher total precipitation, more cumulative snow fall, and a greater proportion of winter (first and fourth quarter) precipitation than other communities; 6) pinyon-juniper sites have a dry second quarter, but mountainbrush sites are dry the third quarter; 7) sites that receive a greater amount of the annual precipitation in the first and second quarters are usually more successfully seeded to cool season grasses. Physical site factors illustrated in Table 2 should be considered in revegetation efforts.

Fire-Related Restoration Factors

Certain practices are universally important to all methods of planting wildland sites. However, when fire is used to control competition and native herbs and shrubs are the principal species planted, particular emphasis must be given to planting conditions. Four major factors must be considered when fire and seeding are used to improve wildland sites:

1. Elimination of existing competition and the natural invasion of desirable and undesirable plants.

2. Availability and features of the planting stock.

3. Seedbed conditions.

4. Methods and timing of planting.

Plant Competition

Shrubs

Fires can be effective in removing or reducing existing vegetation. Fire normally does not destroy woody species that are capable of resprouting: Rocky Mountain maple (Acer glabrum), mountain snowberry (Symphoricarpos oreophilus var. oreophilus), Woods rose (Rosa woodsii), and black common chokecherry (Prunus virginiana var. melanocarpa). However, resprouting is both genetically and environmentally controlled (Wright and Bailey 1982). Fire can be detrimental to some resprouting species including bitter cherry, Martin ceanothus (Ceanothus martinii), Saskatoon serviceberry (Amelanchier alnifolia), and true mountain mahogany (Cercocarpus montanus). Shrubs that are capable of resprouting will normally recover within 10 to 15 years. During initial years of recovery, grasses and broadleaf herbs can become established.

Table 2. Site comparisons among big sagebrush, mountainbrush, and pinyon-juniper communities[1]

	Habitat or Community				
Characteristic	Basin big sagebrush	Wyoming big sagebrush	Mountain big sagebrush	Mountainbrush	Pinyon/Juniper
Frost-free days	147.2±7.5	115.4±9.8	105.2±13.0	102.8±11.1	117.5±6.8
Elevation (ft.)	5,112	5,266	5,812	5,879	6,068
Maximum temperature (annual) (F°)	64.8±0.9	63.8±0.8	59.7± 0.9	60.6± 1.0	62.0±1.2
Minimum temperature (annual) (F°)	36.5±1.0	31.4±1.2	30.0± 1.7	31.2± 1.6	32.6±1.0
Maximum temperature (Jan.-F°)	39.9±1.0	38.6±0.9	36.0± 1.0	36.6± 1.3	38.8±1.6
Minimum temperature (Jan.-F°)	17.2±1.0	12.6±1.1	11.7± 2.6	13.5± 1.7	13.8±0.8
Maximum temperature (July-F°)	91.0±0.9	90.4±1.0	85.0± 1.2	86.9± 1.1	88.0±1.2
Minimum temperature (July-F°)	58.3±1.2	52.7±1.5	49.6± 2.0	50.9± 1.8	53.7±1.4
Precipitation (annual)(inches)	13.8±1.1	10.6±0.7	16.5± 0.7	19.6± 1.1	12.3±0.6
% First quarter	29	25	30	32	25
% Second quarter	25	26	24	23	22
% Third quarter	20	25	20	17	28
% Fourth quarter	26	24	26	26	24
Cumulative snowfall (inches)	46.7±5.9	38.9±4.7	69.8± 5.5	95.5±11.6	55.4±7.8
Estimated wind (miles)	50.4±3.0	55.1±8.8	46.0± 6.6	37.5± 3.4	47.0±7.6
Cumulative solar radiation (langleys)	464.1±11.9	510.0±8.7	480.9±14.3	480.4±12.5	480.5±16.7
Annual daily minimum relative humidity (%)	37 ± 3	26 ± 2	33 ± 3	33 ± 2	33 ± 3
Annual pan evaporation (inches)	66.1±2.0	64.3±4.8	51.6± 2.3	52.2± 2.7	56.8±2.7

[1]Data extracted from Stevens et al. (1983). Units used by Stevens et al. were kept. Values are mean ± standard error of the means. Plant communities were selected by authors' observations and by consulting with Foster (1968). Weather records for communities were as follows: Basin big sagebrush; n=12; Beaver, Blanding, Brigham City, Ephraim, Fillmore, Kanab, Moab, Ogden, Parowan, Pleasant Grove, Scipio, Tooele; latitudinal range, 37°37'--41°31'. Wyoming big sagebrush; n=7; Elberta, Fairfield, Koosharem, Levan, Milford, Modena, Snowville; latitudinal range, 37°48'--41°58'. Mountain big sagebrush; n=8; Alpine, Alton, Coalville, Heber, Kamas, Logan USU, Monticello, Scofield; latitudinal range, 37°15'--41°40'. Mountainbrush; n=8; Bryce Canyon NP, Deer Creek Dam, Eureka, Morgan, Mt. Dell Dam, New Harmony, Snake Creek, Timpanogos Cave; latitudinal range, 37°29'--41°02'. Pinyon-juniper; n=6; Cedar City, Cove Fort, Echo Dam, Escalante, Hiawatha, Tropic; latitudinal range, 37°42'--40°58'.

Fires have a different influence upon non-sprouting shrubs that depend upon seed for reproduction. Seed-dependent plants must produce an abundance of seedlings immediately after the fire to repopulate the area. Seedlings can arise from seeds stored in the soil, or from seed produced by plants remaining after burning. Seeds of many species are usually very tolerant of heat (Daubenmire 1968), and if slightly covered with soil can survive intense fire (Wright and Bailey 1982). Mueggler (1956) reported that sagebrush seedlings usually develop from an existing seedbank, but stand establishment is erratic as burning may kill over 75 percent of the seeds in the soil at or above 1/4 inch depth. Stevens et al. (1981) reported that seeds of certain shrubs including big sagebrush, rubber rabbitbrush (Chrysothamnus nauseosus), and common winterfat (Ceratoides lanata) may not be viable for more than two to three years. These plants do not regularly produce an abundance of seeds due to the arid conditions in which they grow. Consequently, regeneration can be negated by a lack of viable seed. Reburning can have long-lasting effects upon regeneration of plants such as big sagebrush as seed-bearing plants are eliminated and the seed supply is exhausted.

Big sagebrush and other shrub seedlings usually become established the first two years after burning (Blaisdell 1953). Thereafter, seedling numbers tend to diminish due to an increase in competition and reduction in the number of shrub seeds. However, big sagebrush can and will invade dense stands of grass many years after burning (Harniss and Murray 1973).

The size of the area burned and the presence and distribution of unburned sites influence big sagebrush recovery (Mueggler 1956). Johnson and Payne (1968) reported that sagebrush reinvasion results from surviving plants. Large fruits or seeds that are carried by rodents may be displaced long distances from the mother plant. Unless mature plants are uniformally left throughout a burned site, repopulation by natural seeding can be slow. However, livestock grazing can hasten the recovery of woody species (Astroth and Frischknecht 1984).

Herbs

The recovery of native plants and seeding success influence postfire succession. Several annual and perennial herbs reoccur quickly (Everett and Ward 1984) and can suppress the establishment of seeded species. Cheatgrass (Bromus tectorum) is the most influential weedy species in the sagebrush, pinyon-juniper, and mountainbrush communities. It occupies extensive areas and spreads rapidly following disturbances. Sites supporting this weedy grass are closed to the natural establishment of desirable perennials (Piemeisel 1938, Robertson and Pearse 1945). Seedings are usually unsuccessful unless the area is cleared of the weedy competition (Hull and Holmgren 1964).

Fire is not entirely successful in controlling cheatgrass (Wright et al. 1979). Burning conducted when the cheatgrass seeds have matured, but not fallen from the plant, will consume much of the current year's seed crop. Young et al. (1976) have recorded over a 90 percent decrease in seed density through controlled burns. Pechanec and Hull (1945) also report a significant decrease in seed numbers by burning at selected dates. However, fires do not always consume existing seeds (Table 3), and a serious buildup can occur even with repeated burning (Wright and Klemmedson 1965). Cheatgrass is highly opportunistic. A reduction in seed numbers will not reduce overall competition. A few plants can be as competitive as a large number of individuals.

Table 3. Influence of burning dates upon survival of cheatgrass caryopsis.

Location and treatment	Date of burn	Number caryopsis (ft^2) Mean[1]	Standard deviation
Highway 6, Juab Co., UT	June 1984		
Burn		1131	713
Unburned		2669	2124
Desert Mountain, Juab Co., UT	July 1984		
Burn		502	366
Unburned		2746	1976
Yuba Lake, Juab Co., UT	August 1984		
Burn		186	179
Unburned		2895	2433

[1]Means were significantly different ($P < .05$) between treatments at each site by "t" test analysis.

Piemeisel (1951) concluded that the conversion of sagebrush-grass ranges to annual weeds progresses through three distinct communities. Barren areas are first occupied by Russian thistle (Salsola pestifer), then by mustards (Descurainia sophia) and tumblemustard (Sisymbrium altissimum), and finally by cheatgrass. Russian thistle dominates the first two years, mustards the third and fourth, and cheatgrass from the fifth year on. The changes take place regardless of differences in weather. The communities thrive and reproduce within the limits of available moisture. Once cheatgrass has gained control, the sequential changes in community structure are not repeated if the sites are again disrupted. Sustained cheatgrass dominance is critical to direct and natural seedings. Sites can be much more successfully seeded when Russian thistle or mustards are present than when cheatgrass has assumed control.

Russian thistle initially spreads over unoccupied ground. It matures seed late in the summer, but has no fall regrowth. Seeds normally do not germinate until late spring and maximum growth does not occur until daytime temperatures become quite warm. Cool-season grasses are thus able to compete very well with this annual. Annual mustards germinate in the fall and begin growth earlier in the spring and grow much faster than Russian thistle. However, both introduced and native grasses can compete quite favorably with the annual mustards.

Piemeisel (1938) also determined that development of a cheatgrass community progresses from few individuals, to clusters, and finally to dense patches. The increase in density takes place in designated islands. Different stages are intermixed throughout large areas, and account for differences in plant numbers and ground cover. Mack and Pyke (1984) also reported considerable variation in plant density among sites. This variation tends to facilitate natural or artificial seeding of other species. When cheatgrass numbers are low, other species are more likely to establish. Burning cheatgrass sites, however, may actually increase competition by this annual. Burning tends to thin the stand producing more uniformity and eliminating the islands with varying densities.

Quality of Planting Stock

Most cultivars or agronomic grasses that are commercially produced and seeded on rangelands have evolved through extensive breeding and selection processes. Seeds are normally produced from cultivated fields where the parental plants are maintained under uniform and healthy conditions. Seeds produced under these circumstances usually germinate uniformly and predictably.

Most grasses and broadleaf herbs used in rangeland seedings are easily germinated and establish very well. Species not easily established usually are not planted. Western wheatgrass (Agropyron smithii) and Russian wildrye (Psathyrostachys juncea) are often excluded from seed mixtures because of this weakness.

Seeds of most native herbs and shrubs used in rangeland plantings currently are collected from wildland stands where rearing conditions usually are less favorable. Considerable diversity in seed size, maturation, and germination occurs in these collections (Gross 1984). Quinn and Colosi (1977) concluded that differences in seed behavior are also genetically controlled. Harper et al. (1970) stated that seed size and shape may be genetically regulated, but reported seed dormancy and germination behavior can be affected by the phenomenon of somatic polymorphism. In such cases, seeds of different size, shape, or weight may be produced on the same plant, but will germinate differently. Seeds may also mature at different dates, thus extending the period of germination. These mechanisms may serve to ensure the survival of a species under natural conditions, but may not result in a uniform stand if seeds are planted in controlled conditions.

The growing conditions of most wildland stands cannot be easily altered, yet seed production can be improved through harvesting techniques and management. Wildland seed collectors and land managers can ensure the use of better quality seed through recognition of the factors influencing floral formation and seed production.

Pruning and thinning can improve seed quality and yields. Exposure of seeds or fruits to different photoperiods is known to influence permeability of the seed coat, viability of stored seeds, and germination behavior (Mayer and Poljakoff-Mayber 1982). The effects of light upon seed development of most native species are not fully known. However, seeds collected from certain wildland populations occurring at various elevations and exposures are known to germinate differently. . Land managers are frequently encourage to utilize seed collections made near the proposed planting site to better ensure the use of adapted materials. However, some selections, particularly of Woods rose and skunkbush sumac (Rhus trilobata var. trilobata), are difficult to germinate, yet other sources germinate freely. If adapted, the most germinable seed should be planted.

Harvesting seed from specific sites, at specific dates, or from only particular parts of plants can improve seed quality. The position of the flower on the plant has a major effect upon seed formation and germination. Differences have been reported for certain grasses (Gutterman 1980), and umbells (Thomas et al. 1979).

Seeds of mountain lupine (Lupinus alpestris) and Utah sweetvetch (Hedysarum boreale var. germinale) appear to develop, ripen, and germinate in a sequence according to the position of the seeds in the pod. Seed maturation or development of arrowleaf balsamroot (Balsamorhiza sagittata) proceeds in a definite pattern based on floral position within the head. A higher percentage of filled utricles of fourwing saltbush (Atriplex canescens) tends to occur on the central part of the flower stalk.

Seeds of Apache-plume (Fallugia paradoxa) mature indeterminantly throughout the growing season. Some bushes produce only pistilate flowers; others produce staminate flowers or a combination of both sexes (Blauer et al., 1975). Seed quality varies among bushes and harvesting dates. Seed yields of fourwing saltbush fluctuate annually. Plants are usually dioecious, yet when subjected to stress some of the previously female plants change sex (McArthur 1977).

Utricle size is an important consideration in seeding and in the seed quality of fourwing saltbush (Gamrath 1972, Crofts 1977). Uniform utricle size is desirable in mechanical planting. Smaller utricles generally fill and germinate better than larger ones (Gamrath 1972, Crofts 1977). Some data from individual plants involved in the selection of 'Rincon' fourwing saltbush showed a trend for good quality in smaller utricles (Table 4). However, in this case, differences among utricle sizes were not significant. About 50 percent of filled utricles actually germinate (Table 4). For best quality only mature utricles should be harvested (Crofts 1977).

Collectors can significantly improve seed quality by harvesting only high-quality fruits. The better quality seeds of antelope bitterbrush, Stansbury cliffrose, and true mountain mahogany (Cercocarpus montanus) detatch first from the bush. Immature and poor quality seeds adhere to the plant and disseminate last. Species of Aster, Penstemon, and Eriogonum usually produce an abundance of flowers and fruits, but the number of seeds that mature varies annually. Seed quality and maturation are influenced by pollination, insects, and climate.

Table 4. Utricle fill and seed germination of select bushes of one ecotype of fourwing saltbush.

Seed size	% Filled utricles	% Germination	% Filled utricles to germinate
LARGE (bush) (>11 mm)[1]			
2-34	78	35	44.87
4-21	59	11	18.64
5-20	58	38	65.52
7-63	77	46	59.74
9-13	37	33	89.19
Mean[2]	61.8%	32.6%	52.75
SMALL (bush) (< 7 mm)			
2-5	52	46	88.46
6-4	81	38	46.91
7-49	40	13	32.50
8-32	80	61	76.25
12-52	82	38	46.34
Mean	67.0%	39.2%	58.51
MEDIUM (bush) (9-11 mm)			
3-68	64	37	57.81
4-38	79	41	51.90
5-28	38	21	55.26
8-39	78	46	58.97
12-29	69	43	62.32
Mean	65.6%	37.6%	57.32
Grand Mean	64.8%	36.5%	56.33

[1] Utricle length.
[2] None of the comparisons were significantly different as determined by analysis of variance.

Allowing seeds to properly ripen on the plant can also affect germination. Exposure of certain nongerminating seeds to high humidity can improve germination (Palevitch et al. 1980). Although not adequately tested, seed maturation appears critical to various species that mature in late summer and fall, including winterfat and big sagebrush. Although seeds of winterfat may be easily detached soon after the fruits attain a normal size, seeds left to ripen on the plant germinate more uniformly than those collected early.

Mayer and Poljakoff-Mayber (1982) state, "it is likely that the conditions prevailing during seed maturation are to some extent predictive of conditions which will prevail immediately after seeds are shed. The positional effects are probably yet another way to ensuring the spread of germination over time." Assuming this concept is correct, seeds harvested and planted the same year would likely establish better than seeds stored and seeded in later years. This concept may not be correct, yet some seed collections produce satisfactory stands under specific conditions.

Seeds that germinate and establish universally well should be promoted and utilized. Seeding a wide array of native species may be desirable, yet species that have proven to be most reliable should be emphasized. Certain wildland sites consistently produce an abundance of high-quality seed. Germination features and seedling vigor also remain consistent among years of collection. Seed collectors and land managers should exploit these selections. In addition, seed quality standards should be developed to better regulate the use of wildland seeds.

Effects of Fire Upon Seedbeds

Soil Physical Properties

Wells et al. (1979) concluded that fire influences on soil physical properties and erosion depend upon the intensity of the fire, proportion of the vegetation and litter consumed, heating of the soil, proportion of the area burned, and fire frequency. Most studies indicate that fires are not intense enough to directly affect soil structure unless all litter and vegetation is removed. Altered surfaces may then result in puddling and baking (Dyrness and Youngberg 1957).

Eckert et al. (1978) recognized that different kinds of soil surfaces occur on arid and semiarid ranges. Noncrusted, nonvesicular coppice soils usually appear beneath shrubs and a crusted, vesicular soil occurs between shrubs. The coppice soils are more conducive to seedling establishment having a difference in polygon morphology, vescularity, organic matter, and crust hardness. Seedbed conditions can be seriously affected by physical disruption of the soil surface.

Although soils may not be seriously altered by burning, significant changes can occur to decrease quality of seedbeds. Reduction in infiltration (Buckhouse and Gifford 1976), moderate erosion and runoff (Arend 1941), and loss of foliage and surface litter increase evapotranspiration (Anderson 1976) and moisture availability. Lack of surface protection and loss of soil aggregation increase soil movement resulting in an unstable seedbed. Planted seeds are often exposed or deeply covered by soil movement.

Litter and soil organic matter are important in maintaining soil structure. If soil organic matter is not consumed by burning, there is less change in infiltration and soil erosion. More important, standing litter and plant mass enhance seedbed conditions. The material traps and anchors seeds, reduces wind erosion, and moderates extremes in surface temperature and moisture.

Only a small amount of ash remains following burning. The material is usually blown off site and collects in small restricted areas. Broadcast plantings are often made immediately after a burn in an attempt to use the ash to cover the seed. This is not a wise practice. Few seeds are actually covered, and plantings are often conducted in the wrong season.

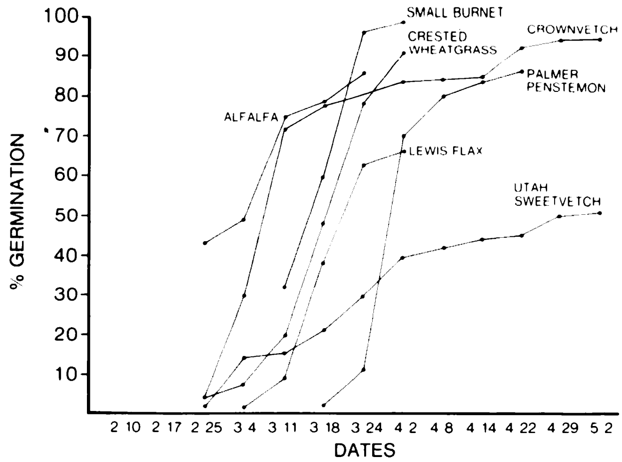
Broadcast or drill seeding of burned sites should be delayed until late fall or early winter when soils have settled following a series of rainstorms. Wright et al. (1976) found that soil losses and adverse surface conditions may persist for 15 to 30 months after burning juniper sites in central Texas. Seeding loose seedbeds is hazardous as planting depths cannot be correctly maintained. Loose soils can be firmed using press wheels attached to the seed drill. Eckert et al. (1978) found deep-furrow drilling to be effective in planting coppice soils.

Soil Moisture and Temperature

Extremely short periods of favorable soil moisture and temperature conditions often occur in either the fall or spring months in arid or semiarid rangelands. Soil moisture, particularly surface moisture, is quickly lost in the early spring. Thus, the rate and period of germination is critical to plant establishment. Seeds that germinate extremely early in the spring may be subjected to losses from frost. Daytime temperatures frequently become warm enough to instigate seed germination, however, reoccurring periods of frost can damage the new seedlings. Seedlings of common winterfat, curlleaf cercocarpus, and big sagebrush often are damaged by early frosts. In contrast, late-developing seeds often fail to become adequately established before soil moisture is depleted. The germination pattern of crested wheatgrass (Agropyron desertorum) serves as a standard to compare with other species (Fig. 1). Plants germinating before wheatgrass are subjected to frost damage. Those that germinate similarly to the grass are usually the most successful species to establish by direct seeding.

Species that germinate even over a short period require frequent rewetting of the soil to maintain small seedlings. Wester and Dahl (1983) determined that alfalfa seeds germinated in three days at temperatures of 86°F, 75°F, or 65°F with one application of 0.2 inches of water. At the two highest temperatures, plants

Fig. 1. Field germination of select broadleaf forbs.



required rewatering within one or two days after emergence or they died. Even at the lower temperatures, plants required rewatering within three days after emergence. Subsequent rewatering is required to maintain small seedlings until they are fully established.

Seedings of species that can germinate in a relatively short interval are often quite successful. Different ecotypes of fourwing saltbush express not only difference in the number of seeds that germinate, but also in the duration of germination (Fig. 2). Planting slowly germinating seeds is not advisable in arid regions or rapidly drying soils. However, seedbed conditions are difficult to predict, and vary from year to year. This is seen in quite different germination patterns of forage kochia (Kochia prostrata) from four different years when planted at the same site (Fig. 3).

Fig. 2. Field germination of select fourwing saltbush ecotypes.

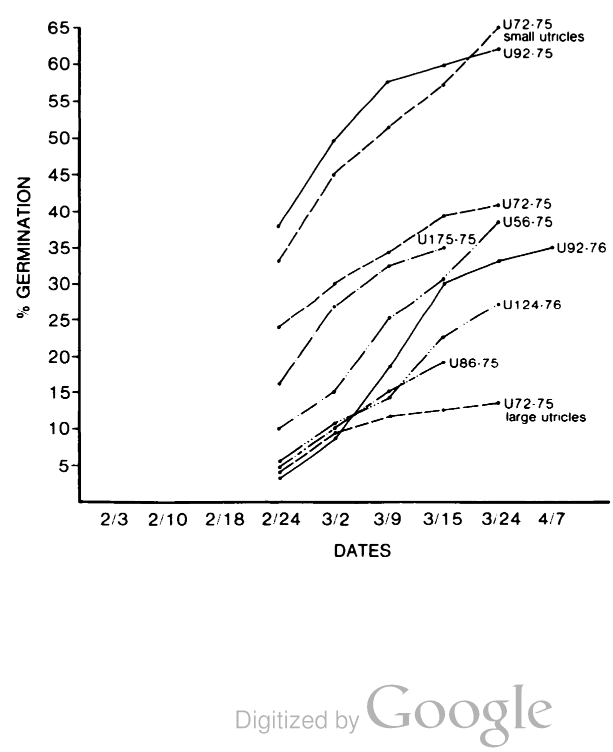
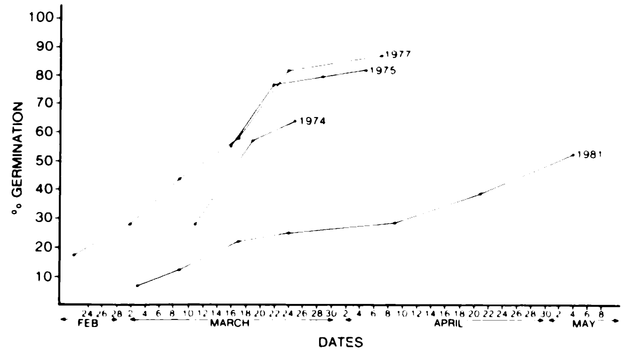


Fig. 3. Field germination of "immigrant" kochia for four different years Ephraim, Utah.



Most shrub and herb species require different periods of cold-moist stratification to overcome dormancy (Table 5). Fall seeding is recommended for species that require long periods of stratification or vernalization (Frischknecht 1959). Nondormant seeds including alfalfa (Medicago sativa), common winterfat, and certain selections of fourwing saltbush can be planted in spring.

Different populations of the same species demonstrate adaptations in their germination requirements in relation to the temperature conditions of the collection site (Barclay and Crawford 1984). Although not adequately confirmed, it has been observed that separate collections of bluebunch wheatgrass (Agropyron spicatum), bottlebrush squirreltail (Sitanion hystrix), and western yarrow (Achillea millefolium var. lanulosa) germinate differently. Whether these differences are significant and may affect Planting success is not known, but they appear to be important.

The range of soil temperature at the time of germination can affect the group of plants that germinate. When soils are moistened at low temperatures chiefly winter annuals, e.g. cheatgrass, germinate. Soils moistened at higher temperatures tend to produce summer annuals (e.g. Russian thistle). Extreme differences in soil temperature occur from year to year in arid and semiarid regions, and quite different groups of plants appear.

Burns increase soil temperatures in grasslands by 19° to 26°F (Weaver and Rowland 1952). Sharrow and Wright (1977) recorded increases of 20 to 21°F. The increase is due to absorption of solar radiation by blackened soils unshaded by litter and live plants. The elevated temperatures persist for two years (Hobbs and Schimel 1984). Vegetation on burned areas thus begins growth two to three weeks earlier than nonburned sites (Ehrenreich and Aikman 1963).

Table 5. Seed germination features of selected shrubs.

Common name	Percent germination[1]	Stratification period (days)	Germination period (days)
Bitterbrush, antelope	90	30	14[1]
Ceanothus, Martin	95	60-90	14
Chokecherry, black	80	60-90	20
Cliffrose, Stansbury	90	30	14
Ephedra, green	90	30	15
Cercocarpus, curlleaf	80	30-90	25
Rose, Woods	75	60-120	30
Sagebrush, basin big	80	Ripening	10
Sagebrush, mountain	70	Ripening	14
Sagebrush, Wyoming	80	Ripening	10
Saltbush, fourwing	40	Ripening	14
Serviceberry, Saskatoon	85	120-180	30
Snowberry, mountain	75	100-200	25
Winterfat, common	80	--	4

[1]Germination recorded under nusery conditions, Lucky Peak Nursery, Boise, ID.

Elevated temperatures also result in a rapid decrease in soil moisture. Diminution of moisture content of the upper layer of soil following fire has been reported for different regions (Haines 1926, Blaisdell 1953, Wells et al. 1979). Blaisdell (1953) found that reduction in moisture content from burning sagebrush ranges was only temporary, and that intense fires resulted in a greater decrease of soil moisture than light or moderate fires. Sharrow and Wright (1977) concluded that reduced soil moisture on burned areas is primarily due to increased transpiration by the rapidly growing plants. However, Whisenant et al. (1984) found that soil moisture reduction from burning of cool-season grasslands resulted from increased evaporation from the soil and greater transpiration by the remaining plants.

The initial decrease in soil moisture that occurs following burning is critical to seedling establishment. Maintaining an adequate amount of soil moisture in the surface soil horizons is essential to seed germination and seedling establishment. If possible, litter should be retained to protect the soil surface. If the litter has been consumed by burning,. shallow, but stable, furrows can be used to moderate rapid drying and extreme seedbed temperatures. Harrowing the soil surface can create a surface mulch to reduce loss of surface soil moisture.

Soil Nutrients

Hobbs and Schimel (1984) concluded that fire in the sagebrush-grassland and mountain-brush communities increased the rate of N mineralization and presumably the availability of N to the vegetation. An increase was recorded for one year on sagebrush sites and two years in the mountainbrush sites. Nitrogen losses occurred from erosion and may not have been compensated by N fixation. Wells et al. (1979) reported that nitrogen fixation, both symbiotic and nonsymbiotic was more prevalent following fire, with general increases in P, K, Ca, and Mg. Nutritive imbalance or changes in soil pH have not been observed to adversely affect seedling establishment or significantly improve it.

Effects of Light

Seed response to light is important in semiarid rangeland seedings as many small-seeded species are often planted and planting depths are difficult to regulate. Seeds may require light to germinate, be inhibited by light, or be indifferent to light. No large seeds are known to require light, but many small seeds do. Small seeds have little reserve and require early photosynthesis (Mayer and Poljakoff-Mayber 1982). Planting small seeds at too great a depth is a major cause of seeding failures. Loose soils, sometimes caused by burning cannot be easily planted and seeds are often Placed too deep or too shallow. Difficulties have been experienced when seeding sagebrush or forage kochia on burned sites. Seeds require a moist seedbed, created by a light coverage of soil, but cannot survive deep planting or barren surface conditions.

Larger seeds are less sensitive to planting depths, and are much easier to seed. Small seeds are more successfully established by broadcast planting than deep furrow drilling. Monsen and Richardson (1983) found that small seeds can be successfully planted using the Brillion cultipack seeder.[1] The machine is commonly used to plant lawn grass and other very small seeds. Broadcast seeding can be conducted by dispensing seed in front of a drill or before chains or other implements are used to "rough" or slightly disturb the soil surface.

[1] Use of trade or firm names is for information only and does not imply endorsement by the U.S. Department of Agriculture of specific products or services.

Seed Features

The position of a seed in the soil, as well as planting depth, has a significant effect upon germination. Stevens et al. (in press) found that rubber rabbitbrush seeds germinate and survive better if positioned upright in the soil. Also, seeds with attached pappus survived much better than cleaned seeds or those with detached pappus. The fruits of many Compositae bear a pappus that causes the basal attachment scar to be positioned on or in the soil. Soil moisture uptake occurs through fissures or points at the micropyle (Sheldon 1977). Booth and Schumann (1983) found that up to 24 percent of cleaned seeds (utricle and bracts removed) of common winterfat did not show positive geotropism and failed to properly establish. The rate of water uptake is dependent, in part, upon contact with the soil. Small seeds have large contact surface areas that result in rapid uptake of soil moisture.

Peart (1984) discovered that seedlings of passive awned grass species (awns that do not twist or turn) arose either from seeds that were found in standing position with the calloused ends anchored in the soil or from seeds that lay unanchored and almost horizontal. Actively awned seeds germinated from buried seeds. Unawned seeds were found to germinate either from seeds lying on the surface or buried to a depth of 1/8 inch. Seedlings from most buried seeds arose in small clusters. Regardless of planting position, seeds must become anchored to survive.

The presence and abundance of mucilage or appendages affect the uptake of water by seeds, thus regulating germination. Seeds of serviceberry, squaw-apple (*Peraphyllum ramosissimum*), and species of *Penstemon* develop a sticky mucilage when germinated in a wet medium. Molds and fungi frequently grow on the material and damage the germinating seeds.

Seed germination is frequently regulated by different mechanisms, apparently to assure survival of the species. Growth inhibitors may be contained in the fruits, seeds, or leaf litter (Mayer and Poljakoff-Mayber 1982). In arid communities many important species have seed inhibitors. Seed germination of various species of *Atriplex* can be influenced by compounds contained in the seed appendages. Common wormwood (*Artemisia absinthium*) contains a germination inhibitor that is detrimental to the establishment of some plants including *Senecio*, *Lathyrus*, and *Linum* (Mayer and Poljakoff-Mayber 1982).

The extent to which germination inhibitors may affect range seedings is not fully understood. Seed dormancy and hard seedcoats are evident problems with certain important species. Many shrubs that are principal species of the mountainbrush communities are difficult to establish due to these problems. Of principal concern are: Saskatoon serviceberry, black chokecherry, bitter cherry, skunkbush sumac, Woods rose, and mountain snowberry. Seeds of these species must be scarified and planted in the fall to attain the best germination.

Most seeding devices are not designed to position seeds in the furrow, or plant a cluster of seeds in a small group. However, a high percentage of planted seeds need not establish to achieve a desirable stand. Attempting to position different seeds simultaneously with a single drill or seeder may not be feasible, and dispensing seeds with different equipment is often necessary to develop mixed communities. Seeding certain species with a conventional drill, and interseeding select seeds with more specialized equipment is a practical approach. The Hansen seed dribbler is a useful machine for interseeding specific seeds. Other specialized machines are also available (Brown and Hallman 1984).

Timing of Planting

Two planting periods--fall and spring--are recommended for seeding arid and semiarid sites. Spring plantings are much less likely to succeed unless nondormant and easily germinating species are seeded. Hyder et al. (1955) considered drought and improper seed coverage as the two common causes of seeding failures. Plummer et al. (1968) stated that moisture is the critical factor regulating planting success, and emphasized the need to seed in the fall.

Planting out-of-season is a principal cause of seeding failures, and should be strictly avoided. Planting regulations that require seeding to be completed within specified periods following burning are often unrealistic. Seeding dates should be based upon seasonal conditions. Planting in the late summer (August-September) is too early for the lower elevation sagebrush ranges. Seedings should be delayed until late fall or early winter. Seeding during the midwinter months is usually possible in the lower rangelands and is much more successful than summer planting.

Planting Recommendations

The following recommendations should be considered when planting native herbs and shrubs. Most apply to any seeding program, including burned sites.

1. Acquire and plant only high-quality seed. Emphasize the collection and use of seed from wildland sites that produce good-quality seed.

2. Utilize only proven species.

3. Plant the seed at the proper depth. When seed mixtures are used, different planting depths may be required.

4. Plant incompatible species in separate rows.

5. Control competition, retain surface litter, and employ other practices to reduce the loss of soil moisture.

6. Plant in the proper season.

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